
# ASSESSING THE RISK OF POLLUTION FROM HISTORIC COASTAL LANDFILLS

JAMES HAMILTON BRAND

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**Submitted in partial fulfilment of the requirements**
**of the Degree of Doctor of Philosophy**

# STATEMENT OF ORIGINALITY

I, James Hamilton Brand, confirm that the research included within this thesis is my own work or that where it has been carried out in collaboration with, or supported by others, that this is duly acknowledged below and my contribution indicated. Previously published material is also acknowledged below.

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Date: 3rd February 2017

Details of collaboration and publications:

EM31 surveys were undertaken with the assistance of Professor Andrew Cundy and Dr. Martin Smith from the University of Brighton. Data were downloaded and processed by Dr. Smith who also advised upon data interpretation.

Structural Soils Limited undertook the windowless sampling, and Opus International Consultants (UK) excavated the trial pits and assisted with sample collection. The author selected the sampling locations and samples.

Analysis of the screening samples of waste for metal contaminants was undertaken by the EA National Laboratory. All PAH analyses were undertaken by the EA National Laboratory. ICP-MS analyses were undertaken by Dr. Robert Clough at the University of Plymouth. All data analyses and interpretations were carried out by the author.

This research was financially supported by the Environment Agency and Southend Borough Council.

## Abstract

In England, 1264 historic landfills are in coastal and estuarine locations that are low-lying and at risk of flooding and/or erosion if flood defences are not adequately maintained. With increases in sea level, extreme weather events and coastal erosion predicted due to climate change, it is increasingly likely that these landfills will be inundated or breached, which could result in the release of contaminants through leaching or direct release of waste into the intertidal zone. Prior research has focused on the risk of pollution from landfill leachates under normal operating conditions, i.e. waste is fully contained and the landfill is not flooded. This is the first research to assess the risk of estuarine and coastal pollution in the event of historic coastal landfills in England being inundated or waste eroding from them.

An investigation of two landfills in Essex has found that contaminant concentrations in a variety of solid waste materials exceeded sediment quality guidelines, indicating there is potential for adverse effects on flora and fauna if historic landfills erode and waste is incorporated into coastal sediments.

Leaching experiments have demonstrated that seawater flooding of landfills could increase the proportions of metal contaminants released by up to 5,450% compared to freshwater flooding, but adverse effects on surface water quality from leached metals are unlikely for the research sites due to high levels of dilution in the estuary.

The large number of historic coastal landfills, and limited management resources, mean it is necessary to prioritise allocation of remediation funds to sites which pose the greatest pollution risk. Previous methods required extensive data collection to assess the risk of pollution from eroding waste. A new risk screening assessment method is proposed that utilises existing datasets to assess the risk of pollution from historic coastal landfill sites and prioritise them for further investigation/remediation.

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# Chapter 1. Introduction

The need for this research was identified during the development of the Essex and South Suffolk Shoreline Management Plan 2 (Environment Agency, 2010a) when it was recognised that there are flood defences for which a 'hold the line' management policy is being applied solely because there are potentially polluting landfill materials within or behind them when managed realignment may otherwise be the preferred management policy (e.g. Environment Agency, 2012b). The risk of pollution from coastal landfills is not limited to Essex and Suffolk; historically it has been common to dispose of landfill waste within the coastal zone, because marshland was of low economic value and land levels could be raised to reclaim land or prevent flooding (Canning, 1999; Hübner et al., 2010). In England, there are at least 4,748 historic landfills within flood zone 3 and at least 1,264 of them are in areas at risk of tidal flooding (Environment Agency, 2015a; 2015b).

Coastal landfills have been making news headlines in recent years as some have eroded and released waste materials into coastal waters (e.g. NERC, 2008; Pope et al., 2011; BBC News, 2012), but this is not a new problem, landfill erosion has been occurring in Ireland for over 20 years (Belfast Telegraph, 2015) and, with the anticipated effects of climate change, erosion events are likely to become more frequent. With the predicted increases in sea level due to climate change resulting in more frequent coastal flooding (IPCC, 2012), and the anticipated increase in extreme weather events, e.g. more frequent heavy rain storms and fluvial flooding (Environment Agency, 2010a; 2012a), it is increasingly likely that these landfills will be subject to inundation, which could result in the release of contaminants through leaching or direct release of landfill materials as erosion or catastrophic failure of the landfill sites occurs. These problems are especially pertinent to southeast England due to isostatic adjustment and higher relative sea level rise (Environment Agency, 2010a). However, coastal landfills are also an international problem and, in some areas, there are additional climate change pressures, e.g. reduction in sea ice means there is less attention of wave energy exacerbating increases in erosion rates around Alaska (Alaska Department of Environmental Conservation, 2008; Lowe et al., 2009). Whilst there is an abundance of prior research into the risk of pollution from landfills under normal operating conditions, i.e. when the waste is fully contained and the sites are not flooded, only very

limited research has been carried out into the effects of flooding on contaminant release from landfills and this has focused on freshwater not saline flooding (Khoury et al., 2000; Neuhold, 2013). In addition, the risk of pollution from eroded solid waste has rarely been considered, only one published study (Prechthai et al., 2008) measured contaminants in solid waste from a historic landfill site using methods that allow comparison to sediment quality guidelines. Therefore, there has previously been little understanding of the impact of landfills upon the coastal environment both when the landfills are protected from flooding and when the landfills are subject to inundation or erosion (Pope et al., 2011). Consequently, management guidance for such sites has lacked information about assessing the environmental consequences of flooding and erosion. This lack of guidance, combined with legislation that makes it an offence to pollute surface waters, e.g. the Water Framework Directive (Council Directive, 2000), forces coastal managers to choose conservative management policies such as hold-the-line when managed realignment may otherwise be the preferred policy (e.g. Environment Agency, 2012b). Alternatively, where hold-the-line is not practicable, sites are being excavated and the waste materials relocated to mitigate the risk of unknown contaminants being released to the aquatic environment (e.g. Alaska Department of Environmental Conservation, 2008; Alaska Department of Environmental Conservation, 2015). These strategies have huge financial costs, which particularly in the case of 'hold the line' polices make them unsustainable long-term (Nason, 2004). Hence there exists the need for an improved understanding of the contaminants these sites contain and the mobility of the contaminants when waste is inundated in order to understand whether historic coastal landfill sites pose a risk to the aquatic environment. There also exists the need for a method to prioritise which historic coastal landfill sites pose the greatest environmental risk in order that resources can be allocated appropriately for their management (Environment Agency, 2010a; 2012b).

The overall aim of this PhD thesis was to investigate the risk of estuarine and/or coastal pollution in the event of historic coastal landfills being inundated or waste being eroded from them. This was achieved by addressing three research aims and a number of associated objectives. The three aims are stated in the thesis outline below and the objectives are defined in the introductions to chapters 3, 4 and 5.

**Chapter 2** reviews the existing literature relating to the history of landfilling, responsibility for historic landfill management, environmental pressures on landfill sites, landfill contaminants, contaminant mobility, the potential for pollution and landfill risk ranking methods.

**Chapter 3. The aim of this chapter is to determine the potential for contamination of the intertidal zone if solid waste materials are eroded from historic coastal landfills.** It includes introductions to the research sites, details of the sampling locations and methods for analysing solid waste.

**Chapter 4. The aim of this chapter is to determine the potential for contamination of surface waters by metals released from landfill waste that is either inundated by flood water or eroded into surface waters.** This chapter includes details of the leaching methods and the criteria used for assessing the potential for pollution from landfill leachates.

**Chapter 5. The aim of this chapter is to develop a high-level risk screening assessment methodology that focusses on the risk to the intertidal zone and tidal waters from eroding historic coastal landfills.** This chapter reviews existing datasets and proposes a new risk screening assessment to rank historic coastal landfill sites by their relative pollution risk based on their exposure to coastal processes, the vulnerability of the landfill sites and their potential to pollute environmental and ecological sites.

**Chapter 6** reviews the key findings of the research and suggests areas for further research.

# Chapter 2. Literature review

## *2.1 The history of landfill*

### 2.1.1 The origin of coastal landfills

Disposal of waste materials in landfill sites has its origins in Crete in 3000 BC where waste was placed into soil covered pits (Waste Watch, 2004); however, in the US and Europe waste was primarily disposed of by dumping within cities until the 1800s when the link was identified between poor environmental conditions and disease (National Solid Wastes Management Association, 2008). In the UK, the Public Health Act 1875 required householders to store their rubbish in dustbins and made local authorities responsible for the removal and disposal of waste, which led to the development of the modern landfill. It is worth noting that although the term landfill implies filling a hole with waste (landfilling), it is also used to refer to sites constructed by disposing of waste on the ground and covering it (landraising), such sites have often been used for land reclamation. Historically it was common practise in the UK to tip waste directly onto saltmarsh surfaces that were hydraulically connected to coastal waters (Hübner et al., 2009), and globally intertidal zone locations were often selected for waste disposal either due to their low economic value or to raise land levels to reclaim land and/or prevent flooding (e.g. Construction News, 1992; Canning, 1999; New York City Government, 2001; Spector, 2012). Examples of locations where waste has been used to raise land above the floodplain to enable expansion of towns and cities include canal estates in Queensland and Florida, and ports in Manchester, Southampton, Marseille and Singapore (Turner et al., 1991).

In England, landfills within the coastal flood zone are predominantly found clustered around estuaries with major towns and cities, e.g. London (Figure 2.1), Liverpool and Newcastle upon Tyne, although there are also a significant number around rural estuaries in Southeast England between Harwich and Ramsgate (Environment Agency, 2013f). The locations of these sites mean they are often protected by flood defences and in some cases the landfills form part of the flood defences, e.g. Hadleigh Marsh, South Fambridge and Dengie in Essex are locations known to have flood embankments constructed from waste capped with clay.

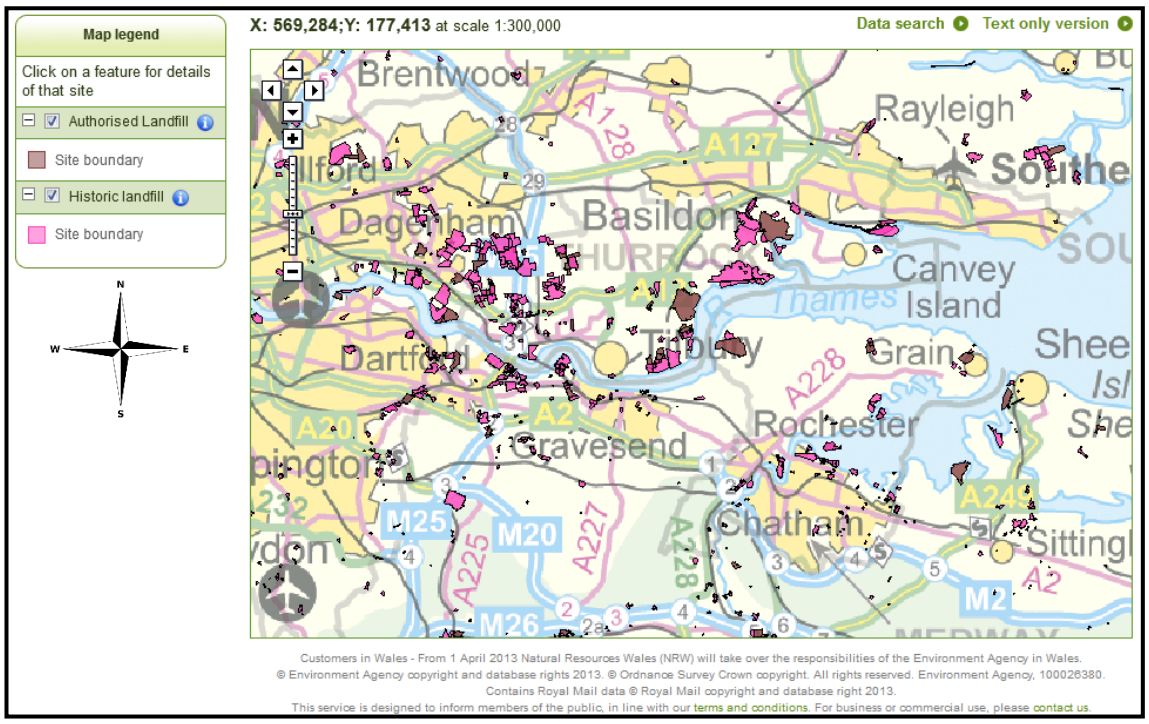



**Figure 2.1: Coastal landfills are typically clustered around estuaries with major towns or cities, e.g. this map shows the abundance of landfills in the Thames Estuary (Environment Agency, 2013f)**

### 2.1.2 The definition of historic coastal landfills and their prevalence

The Construction Industry Research and Information Association's (CIRIA) 'Guidance on the management of landfill sites and land contamination on eroding or low-lying coastlines' (CIRIA C718) (Cooper et al., 2013) estimates there are approximately 1500 historic landfill sites in the 1 in 200 coastal flood zone around England and Wales, and a further 184 that are still authorised to accept waste (Wilkinson, 2009; Cooper et al., 2013). CIRIA's guidance defines historic landfills as those closed before the Waste Management Licensing Regulations 1994, which came into effect on 1st May 1994. However, the Environment Agency's (EA) Historic Landfill GIS dataset guidance note uses the definition: *"A historic (closed) landfill site is one where there is no PPC [Pollution Prevention and Control] permit or waste management licence currently in force. This includes sites that existed before the waste licensing regime, if a site has been licensed in the past, and this licence has been revoked, ceased to exist or surrendered and a certificate of completion has been issued"* (Environment Agency, 2010b). Comparing the May 2013 version of the Historic Landfill GIS dataset with the tidal flooding areas from the EA's May 2013 Flood Map for Planning (Rivers and Sea) -

Flood Zone 3 GIS dataset (using esri ArcMap 10.2), shows there to be 1290 historic landfills in England and Wales with unique Historic Landfill Database Reference numbers which fall at least partially in areas with a 0.5% annual probability of tidal flooding, of which at least 47 closed after the introduction of the Waste Management Licensing Regulations 1994. The discrepancy between CIRIA's guidance and historic coastal landfill numbers identified using the GIS datasets (Table 2.1) is likely to be due to a combination of revisions to flood zones and consolidation of duplicated landfill records. Numbers obtained using either definition are likely to be underestimates as records are believed to be incomplete (Cooper et al., 2013). This research uses the EA definition of historic landfill and the term 'historic coastal landfill' to refer to historic landfills within tidal flooding areas defined in the EA's October 2015 Flood Map for Planning (Rivers and Sea) - Flood Zone 3 GIS dataset. It should be noted that the October 2015 datasets do not include sites in Wales, because since April 2013 the Environment Agency has not been responsible for the management of sites in Wales and the datasets have been updated accordingly. Comparing the October 2015 version of the Historic Landfill dataset with the tidal flood areas from the EA's October 2015 Flood Zone 3 dataset shows there to be 1264 historic landfills in England with unique Historic Landfill Database Reference numbers which fall at least partially in areas with a 0.5% annual probability of tidal flooding (Figure 2.2).

**Table 2.1: Comparison of the numbers of historic landfills present around England and Wales when assessed using different definitions of historic landfill and datasets from different years**

| Historic landfill definition used | Data source | Number of landfills |
|---|---|---|
| CIRIA C718 - closed before the Waste Management Licensing Regulations 1994 | CIRIA C718 published in 2013 | ~1500 |
| CIRIA C718 - closed before the Waste Management Licensing Regulations 1994 | 2013 GIS datasets | 1243 |
| EA - no PPC permit or waste management licence currently in force. | 2013 GIS datasets | 1290 |
| EA - no PPC permit or waste management licence currently in force. | 2015 GIS datasets (England only) | 1264 (England only) |

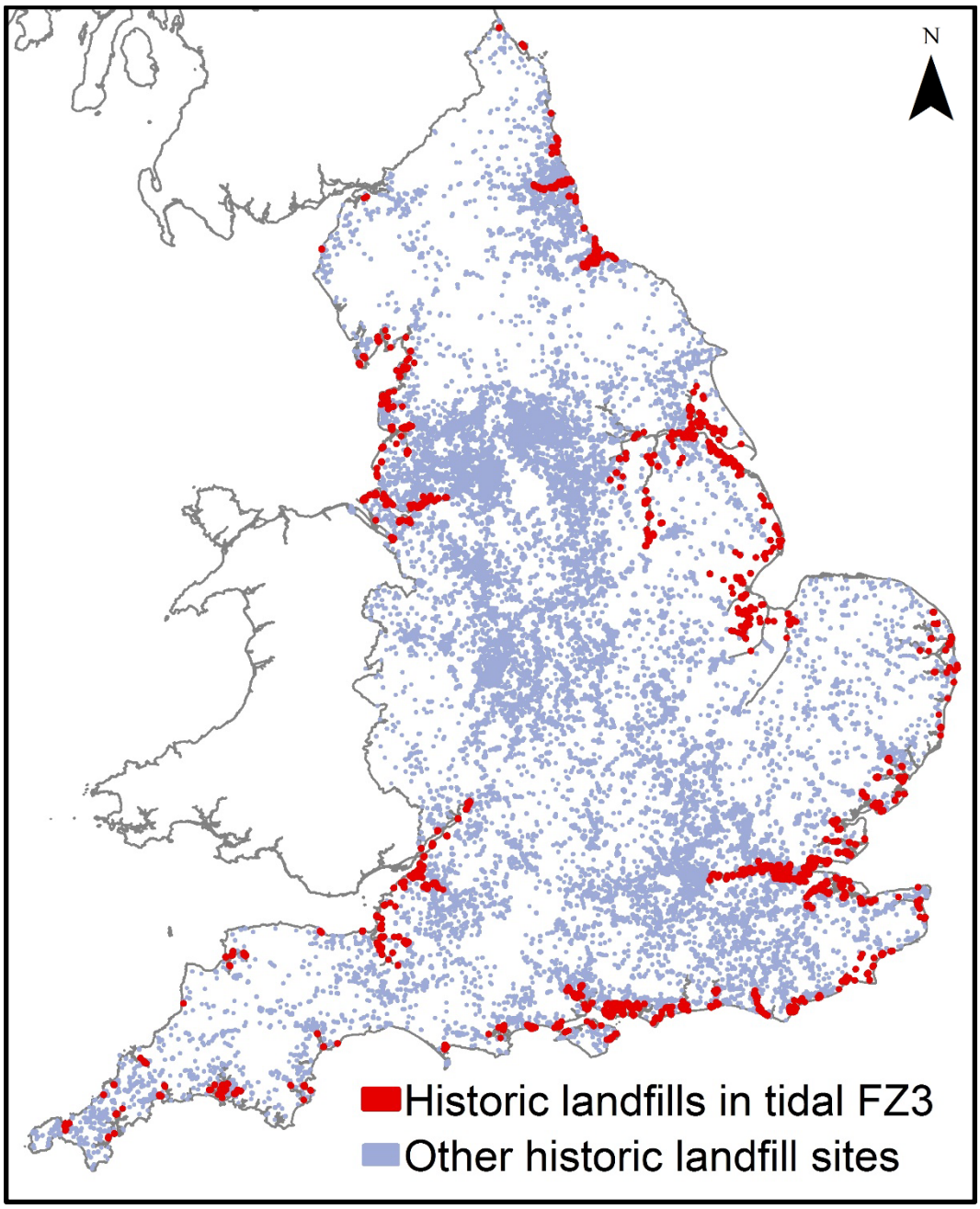



**Figure 2.2: Locations of historic landfill sites in England (not to scale) (Environment Agency, 2015b; Contains Ordnance Survey data © Crown copyright and database right 2016.)**

### 2.1.3 Landfill construction

Historically there have been two types of landfills, the containment type and the natural attenuation type. Natural attenuation landfills relied on the assumption that soil would attenuate all contaminants in leachate before they reached groundwater, but studies suggest that some contamination of groundwater usually occurs (Bagchi, 1983; 1994).

Containment landfills were lined, e.g. with puddled clay, to prevent leachate entering the ground (Bagchi, 1994). Both types are capped using soils upon closure so that the waste is fully enclosed. The Landfill Directive requires that most modern landfill sites are the containment type, a modern site can only be the natural attenuation type if it poses no potential environmental hazard, e.g. it contains only inert waste (Defra, 2010). Coastal landfills are normally protected by flood defences or have raised the land surface sufficiently that they are not subject to regular inundation (Environment Agency, 2013f; Environment Agency, 2013e).

### 2.1.4 Licensing of landfill sites and site records

Typically there are no detailed records of the waste in historic landfills as there were no requirements to keep any records before the licensing of some landfill sites was introduced in the Control of Pollution Act 1974 (Secretary of State, 1974). Only limited records were then required until the introduction of licensing for all landfill sites accepting controlled waste under the Waste Management Licensing Regulations 1994 (Wilkinson, 2012). The Waste Management Licensing Regulations introduced the requirement to keep a record of the location of special wastes within a site and an estimate of the total quantities of biodegradable, non-biodegradable and special waste a site contains (Secretary of State, 1994). 'Controlled waste' is defined as household, industrial and commercial waste or similar waste, and 'special waste' is defined as controlled waste with specific disposal requirements relating to its potential to cause environmental harm (Secretary of State, 1974).

The keeping of more detailed records for landfills has been mandatory since the introduction of The Landfill (England and Wales) Regulations 2002 (Secretary of State, 2002). The regulations require that landfills are classified as hazardous, non-hazardous or inert, and stipulate that hazardous and non-hazardous materials should not be disposed of in the same site, except under specifically defined circumstances. Definitions of the material types are given in the regulations, which also stipulate requirements governing the retention of records relating to waste origin, type, volume and disposal location. The majority of coastal landfills predate the more detailed record requirements introduced in 2002 and records for them only show whether there is evidence of inert, industrial, commercial, household, special or liquids/sludge waste present (see Table 2.2 for definitions), they do not provide details of which materials

were deposited (Environment Agency, 2013f). Consequently, it is often unknown what types and volumes of materials are contained within historic landfills, their distribution within the sites, what contaminants may be present and, therefore, whether they currently pose a pollution risk or will do so in the future if flooding or erosion occurs.

**Table 2.2: Types of waste (source: Environment Agency, 2013f)**

| |
|---|
| **Inert:**<br>Waste which remains largely unaltered once buried such as glass, concrete, bricks, tiles, soil and stones. |
| **Industrial:**<br>Waste from a factory or industrial process. It excludes waste from mines, quarries and agricultural wastes. |
| **Commercial:**<br>Waste from premises used wholly or mainly for trade, business, sport, recreation or entertainment. Excludes household and industrial waste. |
| **Household:**<br>Waste from dwellings of various types including houses, caravans, houseboats, campsites, prisons and wastes from schools, colleges and universities. |
| **Special:**<br>Waste that has hazardous properties and is defined in the Special Waste Regulations 1996. Such properties may be flammable, irritant, toxic, harmful, carcinogenic or corrosive. |
| **Liquids/sludge:**<br>Industrial wastewater, sewage sludge and chemical wastes mixed with municipal solid waste. |

## 2.2 Landfill management and environmental pressures

### 2.2.1 Who is responsible for managing historic coastal landfills?

Identifying who is responsible for the management of contaminated coastal sites that are at risk of flooding or eroding is not always straightforward. CIRIA's guidance for the management of contaminated coastal sites, including landfills, contains a flowchart (Figure 2.3) for identifying the responsible parties (Cooper et al., 2013). However, this does not consider the implications of some landfill sites forming part of the strategic coastal flood defence network, for those sites it is the Environment Agency that is responsible for their management (Defra, 2013c).

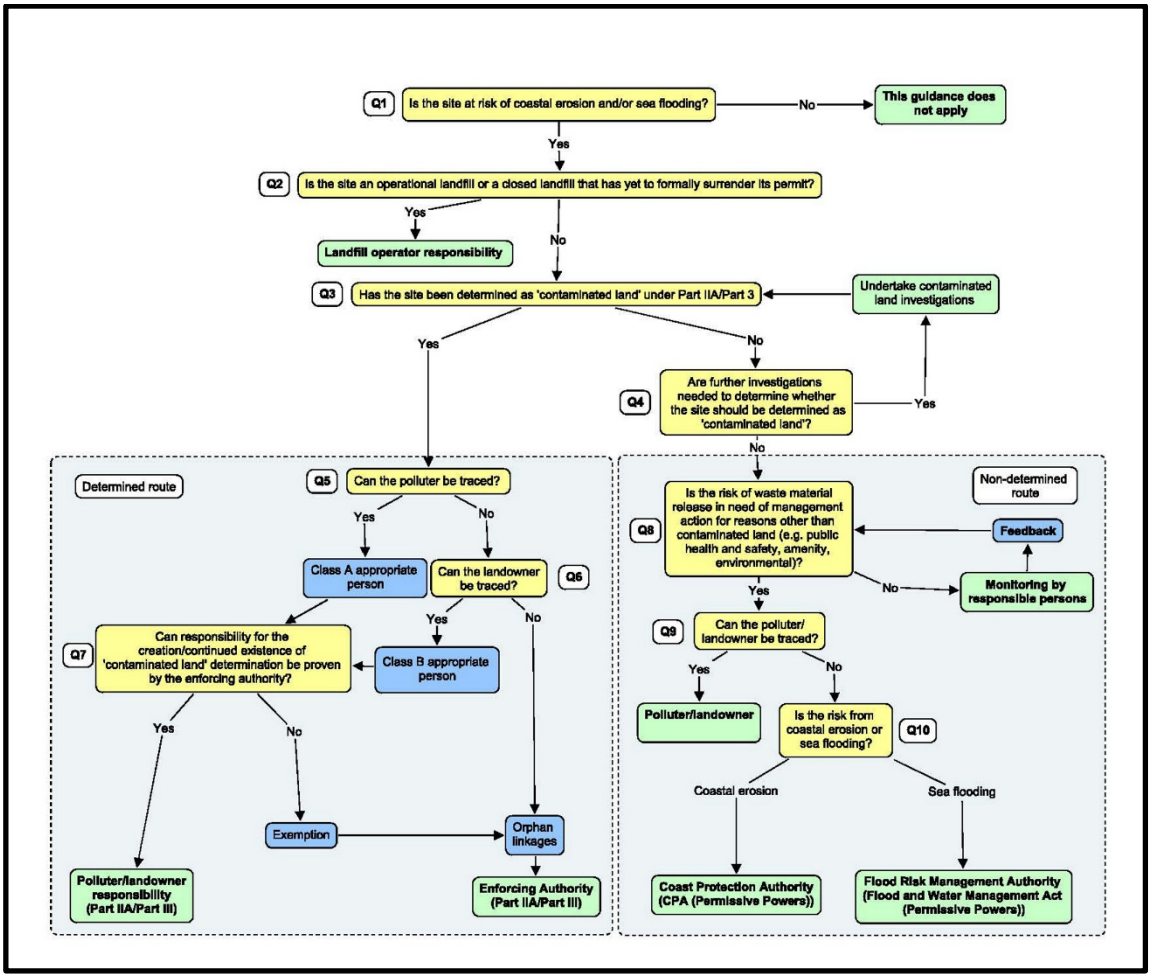



Figure 2.3: Parties responsible for managing contaminated coastal sites (Cooper et al., 2013, p. 10)

### 2.2.2 Coastal management options

There are four categories of management options for coastlines (Environment Agency, 2010a):

1. Hold the Line (HtL), either:
    a. Maintain current defences, i.e. do not make any changes to defences to compensate for sea level rise or glacial-isostatic adjustment of land levels.
    b. Maintain current standard of defence, i.e. increase defence height to compensate for sea level rise and glacial-isostatic adjustment of land levels.
2. Advance the Line (AtL): construct new defences further seaward of the existing defences.
3. Managed Realignment (MR), either remove existing defences completely or breach defences at selected locations, new defences may be constructed further inland or the topography of the area may be such that new defences are not required.
4. No Active Intervention (NAI), cease maintaining defences and allow them to potentially fail, which can result in unmanaged realignment.

The presence of potentially contaminated waste materials within coastal landfill sites and legislation protecting surface waters, e.g. the Water Framework Directive (Council Directive, 2000) and the Water Resources Act (Secretary of State, 2009), restrict which of these management strategies can be implemented. Unless coastal managers can demonstrate that the waste materials pose no environmental risk, coastal landfill sites must be protected by 'hold the line' or 'advance the line' management policies, or alternatively all of the waste must be removed to mitigate any pollution risk. 'Hold the line' is the policy currently being used for coastal landfill sites in England and Wales (e.g. Environment Agency, 2010a; 2012b). In contrast, in Alaska, USA many coastal landfill sites are being excavated and the waste relocated inland (Alaska Department of Environmental Conservation, 2008).

'Hold the line' is unsustainable long-term, both from a nature conservation perspective as it can contribute to the loss of legally protected saltmarsh through coastal squeeze, and from a financial perspective due to restricted flood management budgets especially

when the environmental pressures described in the next section are taken into account (Cooper et al., 2001; Nason, 2004).

### 2.2.3 Environmental pressures on coastal landfills

Climate change is predicted to cause sea level rise, increased saline intrusion into estuaries and rivers, higher coastal storm surges, more frequent and intense storm events and more frequent coastal flooding (Titus et al., 1991; Lowe et al., 2009; Environment Agency, 2010a; 2012a; IPCC, 2012). The increase in flooding frequency will be exacerbated in southeast England by isostatic adjustment causing higher relative sea level rise (Environment Agency, 2010a).

The historic landfills that are the focus of this thesis are in low-lying coastal locations, many of which are expected to suffer from land loss through erosion caused by climate change effects over the next 100 years if remedial actions are not taken (Environment Agency, 2013d). 28% of the English and Welsh coastline is eroding by at least 10 cm per annum (Environment Agency, 2013b). In addition, the presence of flood defences and manmade areas of raised land around the coast restrict natural dynamic coastal processes, and instead of the intertidal zone migrating landwards as sea level rises it is reduced in width, a process known as "coastal squeeze" (Doody, 2013). The intertidal zone, in particular saltmarshes, can significantly decrease the energy of waves passing over it (Möller and Spencer, 2002). Therefore, a reduction in the width of the intertidal zone through erosion or coastal squeeze can result in higher energy waves impacting historic coastal landfill sites or their defences, increasing the probability of inundation through overtopping or site erosion. This could increase leaching as water permeates through the waste or result in the direct release of landfill materials, previously trapped (perched) leachate and contaminated sediments into the wider environment.

Some landfill sites in the UK and elsewhere in the world have already started to erode and release waste, e.g. at Northam Burrows in Devon (BBC News, 2012), Bray near Belfast (Belfast Telegraph, 2015), Nunam Iqua in Alaska (Alaska Department of Environmental Conservation, 2015) and Moreton Island in eastern Australia (Dibben, 2010). It is therefore essential to understand whether waste materials in historic coastal landfills pose an environmental risk if they are allowed to erode and waste materials are released into the intertidal zone. It is also important to understand whether inundation of

historic coastal landfills with increasingly saline waters will affect the contaminant concentrations in the resulting leachates. In addition, given the large number of historic coastal landfill sites and limited funding available for coastal management projects, if historic coastal landfills do pose a pollution risk, it is important to understand which sites are likely to pose the greatest risk in order that they can be prioritised for further investigation and remedial actions. The rest of this chapter will review what is known about contaminants in historic coastal landfills, the effects of salinity on contaminant mobility and existing methods for ranking the environmental risk posed by contaminated sites.

## 2.3 Contaminants in landfill waste

### 2.3.1 Waste composition – deposited materials

Municipal Solid Waste (MSW), which is defined as household or household like waste, includes household collections and some commercial and industrial wastes (Defra, 2013a). Typically there are no detailed records of the waste in historic landfills; however, general records regarding the composition of household waste do exist from the 1930s to 1990s, but these do not detail contaminants present within the materials. The records show that the typical composition of waste has changed during the 20$^{th}$ century as legislation and availability of new materials has changed waste types, e.g. there has been a reduction in ash/dust/screenings and increases in paper and plastics proportions since the mid-1960s (Bridgwater, 1986; Burnley, 2007a). Figure 2.4 shows these changes between the mid-1930s and early 1980s in the UK, and Figure 2.5 shows the typical composition of household waste at the end of the 20$^{th}$ century as determined in a waste strategy planning study for the UK government.

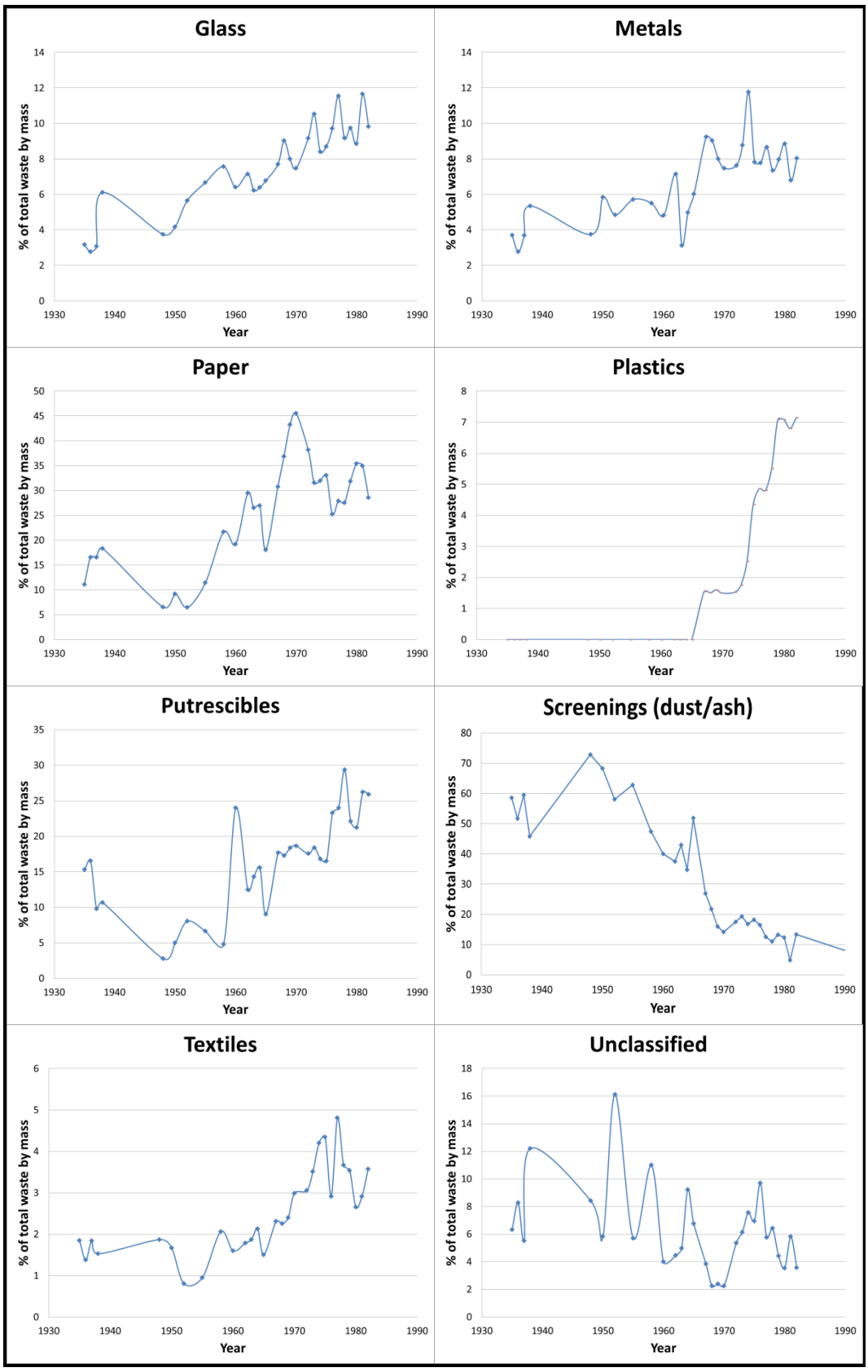



**Figure 2.4: Types of waste by percentage mass in UK household refuse collections between 1935 and 1982 (raw data from Bridgwater, 1986, p. 162)**

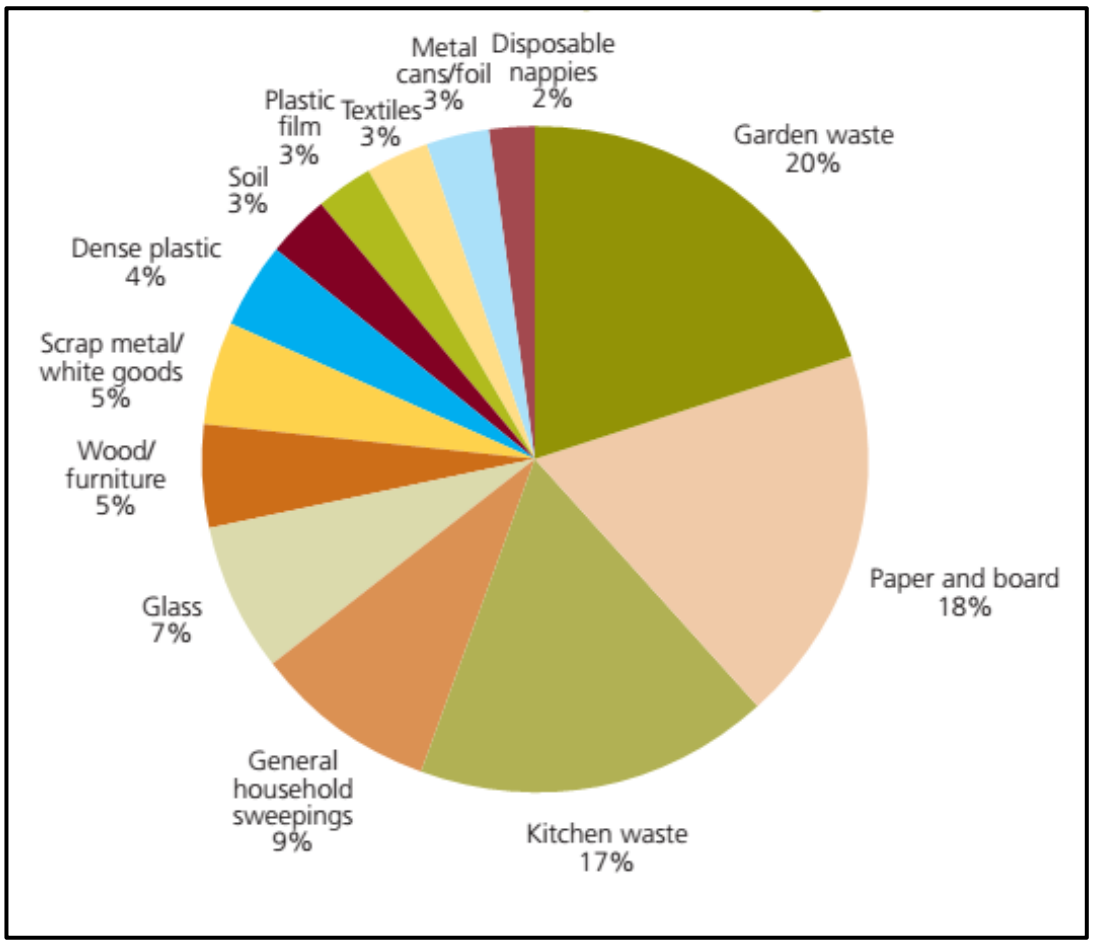



**Figure 2.5: Household waste composition by percentage mass, England (2000/01), as used for UK government waste strategy planning purposes (Parfitt, J. in Defra, 2007, p. 74)**

### 2.3.2 Waste composition – contaminant concentrations

Municipal Solid Waste can contain hazardous substances including cleaning products (acids, alkalis and solvents), batteries (heavy metals, e.g. lead, nickel, cadmium and mercury), pharmaceuticals, pesticides and biocides, oils and fats, paints (solvents and fungicides), wood preservatives (e.g. creosote, tributyltin, and copper chrome arsenate), metal food containers (usually coated with Bisphenol A, an endocrine disruptor) and electrical and electronic equipment (e.g. mercury in fluorescent tubes, heavy metals, CFCs, brominated flame retardants in plastics) (Slack et al., 2004; 2005).

When waste is fully encapsulated there are two pathways for contaminant release: gaseous release and leachate leakage (Bagchi, 1994). Leachate is defined in the Landfill Regulations (Secretary of State, 2002, p. 2) as “any liquid percolating through deposited waste and emitted from or contained within a landfill”. Landfill gas (see Table 2.3 for typical gas composition) is initially the main pathway for emissions, but gas volume

rapidly decreases after closure and in landfills closed for over 20 years it is likely that leachate movement, not gas, will be the main pollutant pathway (Kruempelbeck and Ehrig, 1999). Therefore, studies quantifying contaminant concentrations within the waste have predominately focused upon leachate composition, as leachate movement is generally considered to be the primary pathway for contaminants released from waste. Such studies have either directly sampled landfill leachate or have leached waste samples without measuring the contaminant concentrations in the solid waste or considering the proportion of contaminants that are mobile (e.g. Robinson and Maris, 1979; Robinson et al., 1982; LaGrega et al., 1994; Robinson, 1995; 2007; Ziyang et al., 2009). Leachate is of concern as it may contain elevated concentrations of contaminants, such as organic compounds, Ammonium-nitrogen and heavy metals, and it can also carry suspended solids and insoluble liquids, e.g. oils (Christensen et al., 1994; Robinson, 1995). However, studies focusing on leachate do not consider the scenario where, through erosion of the landfill, solid waste is directly released to the surrounding environment and becomes incorporated into sediments.

**Table 2.3: Typical main components of methanogenic stage landfill gas (Bagchi, 1994).**

| Bulk landfill gas | Proportion (%) |
|---|---|
| Methane | 30-53 |
| Carbon dioxide | 34-51 |
| Nitrogen | 1-21 |
| Oxygen | 1-2 |

One problem with using leachate as a proxy for measuring contaminant concentrations within solid waste is that leachate may not be representative of the landfill as a whole as it can vary between areas of different ages and compositions of waste, and it is common for landfills to contain waste cells of different ages and composition (Kjeldsen et al., 2002). Another shortcoming of using leachate as a proxy for contaminant concentrations in solid waste is that contaminants found in leachate may not be representative of the types and ratios of mobile contaminants within the waste as a whole as studies by Rosqvist and Destouni (2000, p. 56) suggest that "90% of the vertically flowing water moves preferentially through only 47% of the total water content" of the landfill, and as much as 70% of the waste in landfill may not be involved in the leaching process under normal operating conditions (van der Sloot et al.,

2003). In addition, it has been suggested that sampling methods can affect leachate measurements, e.g. heavy metal concentrations in leachate may depend upon the quantity of colloidal matter present and the colloidal content present can be affected by the pumping rate used when taking samples from a monitoring well (Kjeldsen et al., 2002).

A study for the UK Department of the Environment of nearly 4000 leachate samples from 72 domestic waste landfill sites in England, Wales, Scotland and Ireland concluded that there is a profusion of evidence demonstrating that heavy metals (specifically mercury, lead, zinc, cadmium, copper, nickel and chromium) are not commonly present at significant levels in leachates, and that Ammonium-N is the contaminant in leachate that has the highest potential to pollute long-term (Robinson, 1995). This conflicts with Environment Agency (2003b) guidance on leachate management that suggests high levels of heavy metals may be present in leachates. Other researchers agree that although heavy metals are present in leachate they are strongly attenuated within the waste itself by precipitation, complexation and sorption, and are not at concentrations that present a pollution risk, i.e. concentrations in leachates do not normally exceed US or German drinking water standards (Christensen et al., 1994; Kjeldsen et al., 2002). Table 2.4 shows the typical levels of various parameters in municipal waste leachate as compiled for a landfill design handbook by Bagchi (1994). Belevi and Baccini (1990) estimated that 99.9% of metals deposited in landfill waste were retained after about 10 years (mean waste age). UK Red List substances (see Table 2.5) do not pose a serious environmental risk in UK household waste site leachates (Robinson, 1995). However, these studies only consider percolation of leachate into under lying sediments as a pollutant pathway under normal operating conditions, they do not consider the potential for contaminants to be mobilised if landfill sites are flooded or if they erode and release waste materials into the surrounding environment.

**Table 2.4: Typical contaminant concentration ranges in Municipal Waste leachate compiled by Bagchi (1994, p. 52) (ND = no data)**

| Parameter | Range of concentration (mg $L^{-1}$ unless otherwise stated) | Parameter | Range of concentration (mg $L^{-1}$ unless otherwise stated) |
|---|---|---|---|
| Aluminium | ND-85 | Molybdenum | 0.01-1.43 |
| Ammonia-nitrogen | ND-1,200 | Nickel | ND-7.5 |
| Antimony | ND-3.19 | Nitrate-nitrogen | ND-250 |
| Arsenic | ND-70.2 | Nitrite-nitrogen | ND-1.46 |
| Barium | ND-12.5 | pH | 3.7-8.9 units |
| Beryllium | ND-0.36 | Potassium | ND-3,200 |
| BOD | ND-195,000 | Selenium | ND-1.85 |
| Boron | 0.87-13 | Silver | ND-1.96 |
| Cadmium | ND-0.4 | Sodium | 12-6,010 |
| Calcium | 3.0-2,500 | Specific conductance | 480-72,500 μmho $cm^{-1}$ |
| Chloride | 2-11,375 | Sulfate | ND-1,850 |
| Chromium | ND-5.6 | TDS | 584-55,000 |
| COD | 6.6-99,000 | Thallium | ND-0.78 |
| Copper | ND-9.0 | Tin | ND-0.16 |
| Cyanide | ND-6 | TOC | ND-44,000 |
| Hardness | 0.1-225,000 | Total alkalinity | ND-15,050 |
| Iron | ND-4,000 | Total Kjeldahl nitrogen | 2-3,320 |
| Lead | ND-14.2 | Total phosphorus | ND-234 |
| Magnesium | 4.0-780 | Total suspended solids | 2-140,900 |
| Manganese | ND-400 | Turbidity | 40-500 Jackson units |
| Mercury | ND-3.0 | Zinc | ND-731 |

**Table 2.5: Red list substances (The UK Marine SACs Project, n.d.)**

| | | |
|---|---|---|
| 1,2 Dichloroethane | Endosulfan | Malathion |
| Aldrin | Endrin | Mercury |
| Atrazine | Fenitrothion | Simazine |
| Azinphos-methyl | Gamma hexachlorocyclohexane (lindane) | Tributyltin compounds |
| Cadmium | Hexachlorobenzene (HCB) | Trichlorobenzene |
| DDT | Hexachlorobutadiene (HCBD) | Trifluralin |
| Dichlorvos | PCB (Polychlorinated biphenyls) | Triphenyltin compounds |
| Dieldrin | Pentachlorophenol (PCP) | |

Landfill waste composition varies across the world and between sites there are some differences in leachate concentrations relating to differences in landfill age, sampling locations, climate, microbiology, compaction, composition of coverings, and depth of the sites (Robinson et al., 1982; Lisk, 1991). However, several studies have shown that

once landfills sites have reached the methanogenic stage, leachate composition in very large landfills, i.e. >5 $Mm^3$, is similar for a wide range of sites all over the world, (Robinson and Maris, 1979; Rowe, 1995; Robinson, 2007). In the UK methanogenic conditions usually establish within 1-3 years and usually faster in warmer countries (Robinson, 2007), this stage is believed to last for at least a century (Belevi and Baccini, 1990; Robinson, 1995; Kruempelbeck and Ehrig, 1999); accordingly, all historic landfill sites are expected to be in the methanogenic stage and, if over 5 $Mm^3$, have similar leachate compositions.

There are a very limited number of published studies into directly measured contaminant concentrations in solid waste (as opposed to using leachate as a proxy) due to the costs and complexities of obtaining representative results from such highly heterogeneous material (Riber et al., 2007). Comparisons between studies are difficult because they do not use consistent methods of sampling, waste classification or analysis (Burnley, 2007b). Riber *et al*. (2007; 2009) and Eisted and Christensen (2011) sampled household waste MSW at the point of collection, sorted it into material types and digested them using a range of Hydrofluoric Acid (HF) based techniques to determine inorganic contaminant concentrations. The results of these studies into contaminant concentrations in different waste materials are shown in Table 2.6 for elements that are listed in the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life (Canadian Council of Ministers of the Environment, 2002), which are commonly used in the absence of UK sediment quality guidelines. Many of the contaminant concentrations exceed the Interim Sediment Quality Guidelines (ISQGs), below which adverse biological effects are expected to rarely occur, and some of the contaminant concentrations exceed the Probable Effect Levels (PELs), above which adverse biological effects are frequently observed.

**Table 2.6: Comparison of fresh household waste contaminant concentrations to Canadian sediment quality guidelines. Underlined values are below both ISQGs, grey cells exceed at least one of the PELs**

| | As | Cd | Cr | Cu | Hg | Pb | Zn |
|---|---|---|---|---|---|---|---|
| **Canadian sediment quality guidelines (mg kg$^{-1}$)** | | | | | | | |
| **ISQG-freshwater** | 5.9 | 0.6 | 37.3 | 35.7 | 0.17 | 35 | 123 |
| **PEL-freshwater** | 17 | 3.5 | 90 | 197 | 0.486 | 91.3 | 315 |
| **ISQG-marine** | 7.24 | 0.7 | 52.3 | 18.7 | 0.13 | 30.2 | 124 |
| **PEL-marine** | 41.6 | 4.2 | 160 | 108 | 0.7 | 112 | 271 |
| **Concentrations found in waste in Greenland (mg kg$^{-1}$)** | | | | | | | |
| **Paper** | 0.569 | 0.0645 | 7.51 | 50.1 | <0.04 | 2.18 | 17.6 |
| **Cardboard** | <0.3 | 0.0798 | 7.95 | 29.9 | <0.04 | 7.16 | 25.4 |
| **Metal** | 26.9 | 0.22 | 9.37 | 2400 | <0.1 | 79.1 | 18900 |
| **Glass** | 36.5 | 1.42 | 299 | 6.03 | <0.02 | 96.2 | 245 |
| **Plastic** | <0.3 | | 20.9 | 16 | <0.04 | 0.751 | 136 |
| **Wood** | <0.3 | 0.128 | 10 | 32 | 1.8 | 31.7 | 118 |
| **Biowaste** | 3.53 | 0.26 | 1.27 | 8.63 | <0.04 | 0.238 | 34.1 |
| **Combustible** | 0.489 | 1.09 | 177 | 124 | <0.04 | 66 | 192 |
| **Non-combustible** | 132 | 0.77 | 74.5 | 219 | 0.136 | 384 | 361 |
| **Concentrations found in waste in Germany (mg kg$^{-1}$)** | | | | | | | |
| **Fines** | | 0.456 | | | 0.0929 | 75.5 | 112.2 |
| **Organic** | | 0.086 | | | 0.0201 | 8.32 | 404 |
| **Combustible** | | 0.007 | | | 0.0003 | 0.38 | 2.75 |
| **Non-combustible** | | 3.949 | | | 0.0330 | 52.3 | 173 |
| **Paper/cardboard** | | 0.039 | | | 0.0078 | 2.25 | 41.6 |
| **Plastic** | | 0.572 | | | 0.0059 | 12.2 | 56.0 |
| **Textiles** | | 0.045 | | | 0.0067 | 1.56 | 23.1 |
| **Concentrations found in waste in Denmark (mg kg$^{-1}$)** | | | | | | | |
| **Fines** | | | | | | | 18.4 |
| **Organic** | | | | | 0.0175 | 1.18 | 417 |
| **Combustible** | | | | | 0.0072 | 3.69 | 30.5 |
| **Non-combustible** | | | | | 0.0160 | 303.3 | 68.3 |
| **Paper/cardboard** | | 0.023 | | | 0.0137 | 1.55 | 19.3 |
| **Plastic** | | 0.007 | | | 0.0032 | 2.57 | 12.8 |
| **Textiles** | | 0.015 | | | 0.0027 | 4.69 | 6.65 |
| Sources: Greenland (Eisted and Christensen, 2011), Germany and Denmark (Riber et al., 2009) | | | | | | | |

The main disadvantage of using HF based digestions is they extract the residual fraction of metal from the mineral matrix that would not normally be biologically available (Canadian Council of Ministers of the Environment, 2001), which means they can result in overestimating pollution risk when comparing contaminant concentration data to environmental quality guidelines that were developed using other extraction methods, e.g. the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. An example of this is the chromium in glass exceeding all Canadian sediment guidelines (see Table 2.6) when in fact it is likely to be permanently bound within the glass and not bioavailable in the natural environment. This is important when considering the potential for pollution due to the erosion of a landfill site releasing waste to the surrounding environment.

Another method used to obtain contaminant concentration data is elemental analysis of mixed waste samples flash combusted in an incinerator or analysis of slag, dust and gas from MSW incinerators (Brunner and Ernst, 1986; Valencia et al., 2009). Analysis of flash incinerated samples has the same disadvantage as using HF digestions in that the residual fraction of metal that would not normally be biologically available will be extracted. In addition, published papers for analysing the slag, dust and gas from incinerators do not detail how the metal concentrations from these constituents were obtained and, therefore, the robustness of the method is unclear.

As previously discussed, materials entering landfills have changed significantly during the last century, so present day contaminant concentrations in solid waste are unlikely to be representative of contaminant concentrations within historic landfills, even before accounting for a potential reduction in the contaminant concentrations through leaching. Furthermore, chemical and biological transformations of the waste may have resulted in the presence of substances in the waste that were not present when it was deposited (Slack et al., 2005).

Four studies have directly analysed solid waste excavated from landfill sites rather than using leachates as proxies or analysing freshly collected waste. Quaghebeur *et al*. (2013) collected samples of waste landfilled during different time periods and of different waste types from different zones within one landfill. These samples were then sorted and inorganic contaminants in ‘soil type waste’, plastics and paper/cardboard were determined using HF based techniques, with the same limitations described above

regarding comparisons to sediment quality guidelines. Zhao *et al*. (2007) collected waste of differing ages from within a single site and digested mixed waste samples using hydrofluoric acid (HF) based techniques to determine inorganic contaminant concentrations, with the same limitations as described above. Hull *et al*. (2005) analysed fines from a landfill in New Jersey, USA, but did not report the digestion method used. Only one study (Prechthai et al., 2008) has used an acid extraction method designed for determining bioavailable inorganic contaminants (USEPA method 3030B). The results of these four studies are shown in Table 2.7 for elements that are listed in the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Most of the contaminant concentrations exceed the Interim Sediment Quality Guidelines (ISQGs) and many of the contaminant concentrations exceed the Probable Effect Levels (PELs).

**Table 2.7: Comparison of contaminant concentrations in previous studies of excavated solid waste to Canadian sediment quality guidelines. Underlined values are below both ISQGs, grey cells exceed at least one of the PELs**

| | | As | Cd | Cr | Cu | Hg | Pb | Zn |
|---|---|---|---|---|---|---|---|---|
| **Canadian sediment quality guidelines (mg kg$^{-1}$)** | | | | | | | | |
| **ISQG-freshwater** | | 5.9 | 0.6 | 37.3 | 35.7 | 0.17 | 35 | 123 |
| **PEL-freshwater** | | 17 | 3.5 | 90 | 197 | 0.486 | 91.3 | 315 |
| **ISQG-marine** | | 7.24 | 0.7 | 52.3 | 18.7 | 0.13 | 30.2 | 124 |
| **PEL-marine** | | 41.6 | 4.2 | 160 | 108 | 0.7 | 112 | 271 |
| **Concentrations found in landfill sites (mg kg$^{-1}$)** | | | | | | | | |
| **Belgium MSW 'soil', dates landfilled** | **1980-1985** | 61 | 8.5 | 770 | 285 | 2.0 | 500 | 670 |
| | **1985-1990** | 31 | 8.4 | 380 | 205 | 0.50 | 310 | 735 |
| | **1990-1995** | 7.2 | 3.3 | 720 | 760 | 0.27 | 180 | 800 |
| | **1995-2000** | 9.1 | 3.3 | 113 | 107 | 0.19 | 172 | 463 |
| **Belgium IW 'soil', dates landfilled** | **1985-1990** | 19 | 15 | 3800 | 1200 | 2.4 | 1100 | 2600 |
| | **1995-2000** | 22 | 19 | 5730 | 5750 | 0.99 | 2640 | 5600 |
| **Belgium MSW paper/ cardboard, dates landfilled** | **1980-1985** | 39 | 1.4 | 310 | 150 | 0.45 | 440 | 900 |
| | **1985-1990** | 9.7 | 16 | 200 | 210 | 0.27 | 330 | 560 |
| | **1990-1995** | 4.9 | ≤0.40 | 140 | 570 | 0.38 | 54 | 520 |
| | **1995-2000** | 5.5 | 2.7 | 160 | 100 | 0.35 | 380 | 1900 |
| **Belgium MSW plastic, dates landfilled** | **1980-1985** | 33 | 51 | 490 | 1767 | 0.6 | 550 | 1063 |
| | **1985-1990** | 6.9 | 18 | 320 | 150 | 0.36 | 280 | 620 |
| | **1990-1995** | 6.2 | 19 | 280 | 690 | 0.46 | 230 | 1700 |
| | **1995-2000** | 7.6 | 18 | 270 | 270 | 0.1 | 160 | 470 |
| **Belgium IW plastic, dates landfilled** | **1985-1990** | 9.9 | 47 | 780 | 10000 | 2.0 | 1300 | 5500 |
| | **1995-2000** | 12 | 41 | 530 | 2405 | 1.1 | 1900 | 3800 |
| **New Jersey, USA 'fines'** | | | 1.2 | 24 | | | 46 | 406 |
| **South China mixed waste, age** | **2 years** | | 10.7 | 120.6 | | | 171.3 | 556.2 |
| | **4 years** | | 1.1 | 73.5 | | | 134.2 | 549.4 |
| | **6 years** | | 2.7 | 174.6 | | | 141 | 602.1 |
| | **8 years** | | 6.6 | 252.1 | | | 372.2 | 652.9 |
| | **10 years** | | 5.6 | 169.3 | | | 77.9 | 634.2 |
| **Thailand <25 mm fraction** | | | 4.2 | 166.6 | 2245 | | 132 | 1497 |

Sources: Belgium data from Quaghebeur et al. (2013), New Jersey data from Hull et al. (2005), South China data from Zhao et al. (2007) and Thailand data from Prechthai et al. (2008)
MSW = Municipal Solid Waste. IW = Industrial Waste

An alternative approach was taken by Belevi and Baccini (1989; 1990), they considered contaminant concentrations in the waste itself by investigating the proportion of contaminants in leachate samples obtained from landfills against the water soluble contaminants remaining in the landfill waste. Their method involved sampling leachate and leaching waste samples from the same location in order to calculate the proportion of contaminants that had already been mobilised. However, due to the heterogeneity of landfill waste it is unlikely the method used provides accurate data on leachable proportions of contaminants as the leachate is likely to have originated from a greater volume of waste than that analysed, and, due to preferential flow paths forming in waste (Rosqvist and Destouni, 2000), the leachate may not even have passed through the waste tested. In addition, the water extraction method used does not provide contaminant concentration data for the waste that can be compared with environmental quality guidelines for sediment, e.g. the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life require near-total acid extractions to be used (Canadian Council of Ministers of the Environment, 2001), which is important when considering whether waste released to the environment would cause pollution.

In summary, there is a clear need for a study to be undertaken that analyses a range of waste types from historic landfill sites using methods that allow comparison of contaminant concentrations against sediment quality standards/guidelines in order to understand whether the contaminated materials they contain pose an environmental risk if eroded.

### 2.3.3 Landfill decomposition stages and landfill stabilisation

There are a number of stages of MSW decomposition, their effects on leachate production are detailed below:

- The first stage of decomposition lasts a few days or weeks and typically occurs under aerobic conditions. It results in leachates with approximately neutral pH that arise from moisture released through compaction of waste and precipitation (Kjeldsen et al., 2002).
- Anaerobic conditions develop as decomposition progresses. The early stage of anaerobic decomposition (acidogenic/acetogenic phase) produces acidic leachate with high concentrations of soluble degradable organic compounds (Christensen

et al., 1994). During this phase metal concentrations in the leachate peak (Kjeldsen et al., 2002).

- Methanogenic conditions develop after several months or years. Leachate contaminant concentrations drop, but leachates can still have significant amounts of pollutants such as ammonium (Christensen et al., 1994). During this stage leachate pH is close to neutral (Kjeldsen et al., 2002). This is the stage that the majority of historic landfill sites are expected to be at due to the length of time since their construction.
- As decomposition of organic materials nears completion (sometimes called the humic phase), aerobic conditions may return and eventually leachate will become non-hazardous to the environment. This stage is somewhat speculative as landfills have not been studied for a long enough period for it to be confirmed by field studies (Christensen and Kjeldsen, 1995; Kjeldsen et al., 2002).

Landfills are deemed to be stabilised or at Final Storage Quality (FSQ) when they no longer pose a threat to the environment. FSQ is not well defined and there is no general consensus as to how it should be determined (Hjelmar and Hansen, 2005; Valencia et al., 2009). Hjelmar and Hansen (2005) proposed two definitions, the first where leachate meets groundwater and surface water quality criteria, the second where groundwater quality criteria are met at a point downstream of the landfill without using active protection systems. Some researchers (Scharff et al., 2007; Valencia et al., 2009) propose that criteria for FSQ waste should be similar to the Waste Acceptance Criteria (WAC) for inert waste defined in Annex II of the Landfill Directive (Council Directive, 1999; Council Decision, 2003). Other researchers have proposed that landfills should be considered stable when leachate meets United States Environmental Protection Agency limits on contaminants in drinking water (Sizirici and Tansel, 2009). It is worth noting that all of these definitions assume that the waste is permanently contained and can only impact upon the environment via leaching of contaminants, the possibility that a site may erode releasing contaminated waste materials is not considered.

It is unclear from the available literature how long it can be expected to take before leachate can be considered to be non-hazardous. The Environment Agency (2003a; cited in Environment Agency, 2003b) suggests monitoring may be required for 50 years or longer at closed landfills containing biodegradable or other polluting waste. Some

studies support this timescale (e.g. Sizirici and Tansel, 2009), whilst others conclude that leachate from municipal solid waste must be treated for at least 100 years to prevent pollution (Belevi and Baccini, 1990). There is evidence that Ammonium-nitrogen ($NH_4$-N) will be at harmful concentrations for many decades in leachate even from small landfills with high water ingress, and that for large landfills, even when well-managed, leachate could be polluting for centuries (Robinson, 1995; Hall et al., 2006a; 2006b; 2007). Krümpelbeck and Ehrig (1999) compared their findings from a study of German landfills to the findings of other authors (see Table 2.8) and concluded that the average aftercare period (i.e. period for which leachate is still environmentally harmful) will be at least 100 years, with the concentration of $NH_4$-N being the determining factor. The large variation in the findings of different researchers may be due to a variety of reasons, including different landfill structures studied, differing materials in the landfills, different sampling techniques, different modelling techniques and/or different assumptions made by the authors during extrapolation of experimental results. Again it is worth noting that these conclusions do not consider disturbance to the landfill caused by flooding or erosion.

**Table 2.8: Aftercare periods for landfills approximated by different authors, compiled by Kruempelbeck and Ehrig (1999)**

| | Threshold values[1] (mg $L^{-1}$) | Belevi and Baccini (1989) | Kruse (1994) | Heyer *et al.* (1997) | Kruempelbeck and Ehrig (1999) |
|---|---|---|---|---|---|
| **$C_{org}$**<br>**COD** | -<br>200 | 500-1700 years<br>- | -<br>280 years | -<br>80-360 years | -<br>65-320 years |
| **$N_{ges}$**<br>**TKN**<br>**$NH_4$-N** | 70<br>-<br>- | 55-80 years<br>-<br>- | -<br>815 years<br>- | -<br>120-450 years<br>- | -<br>-<br>Decades to centuries |
| **Chloride** | 100 | 100-150 years | 210 years | 90-250 years | 25-130 years |
| **AOH** | 0.5 | - | - | 30-210 years | 40-100 years |
| **Heavy metals** | 0.1-2 | - | - | - | < 10 years |
| [1] From German waste water regulations, except for chloride, which is from Swiss waste water regulations | | | | | |

The main environmental variables controlling stabilisation rates of landfills are moisture content, temperature, available oxygen, refuse particle size and in-place density, nutrient content, microbial state and pH (Noble et al., 1989; Adhikari et al., 2014). It is commonly believed that landfills partially below the water table stabilise much faster than those above it, and that 40-70% moisture content is optimal (see citations in Noble et al., 1989). However, Khoury *et al*. (2000) recognised that previous studies had not looked at seawater intrusion and found in a preliminary study that saline water may actually impede stabilisation due to biodegradation processes being inhibited by increased concentrations of sulfates and chlorides, which may mean landfills in coastal environments pose a pollution risk for even longer if they have hydrological connectivity to saline waters. This deduction was based upon very limited data and Khoury *et al.* (2000) concluded that further research is required into the effects of water salinity on landfill stabilisation.

## 2.3.4 Contaminant mobility and the impacts of climate change

### 2.3.4.1 Contaminant mobility under normal operating conditions

Until a landfill has stabilised the primary source of contamination is the deposited waste material. Under normal operating conditions, i.e. when sites are not flooded and the capping materials are intact, the main long-term impacts of landfill waste are generally recognised to be related to the production and movement of leachate (Hjelmar and Hansen, 2005). Thin layers of soil covering have no effect on leachate production, and

up to 40% of precipitation may become leachate even at landfills with several metres of soil covering (Kruempelbeck and Ehrig, 1999). Recent research has found evidence of contaminant plumes in sediments around historic landfill sites in the Thames Estuary (O'Shea, 2016), and it is likely that other coastal landfills are currently releasing leachate as historic landfills are often unlined, and it is known that leachate will escape even from well-engineered, lined landfills (Arneth et al., 1989; Bagchi, 1994; Allen, 2001).

Leachate plumes can extend over many hectares and their vertical movement can be unpredictable due to the density of the leachate (Bjerg et al., 2011). The shape of the plume is affected by leachate density and groundwater movement (Christensen et al., 1994). Not all leachate will drain to the bottom of the site or flow out of the site, waste absorbs some and some leachate may remain 'perched' if there are layers of low permeability materials, e.g. intermediate soil covers (Bendz et al., 1997; Zhang et al., 2013). A number of factors affect the rate of leachate leakage, these include: the types of material surrounding the waste, the pressure head of leachate, piezometric head around the site and preferential flow paths being present, e.g. holes in lining materials caused by mechanical damage or chemical interaction with waste (Bagchi, 1994; Giroud et al., 1997; Allen, 2001).

Contaminants in leachate may be attenuated by (i) biological action, (ii) cation and anion exchange reactions, (iii) filtration, (iv) precipitation and (v) adsorption as it passes through sediment within the landfill itself or in the surrounding environment, and they may also be attenuated by (vi) dilution (Bagchi, 1987). However, some constituents are not attenuated in the unsaturated zone (e.g. nitrate, chloride and organics) and waters down gradient from landfills, especially those that are unlined, will always be degraded by leachate (Bagchi, 1994; Bjerg et al., 2011). The attenuation processes are considered in more detail below:

(i) Micro-organisms can absorb and breakdown or retain constituents of leachate. Micro-organisms in capped landfills are heterotrophic and as the strength of leachate decreases with time micro-organism activity decreases and contaminants previously fixed in microbial cells could be released as organisms die making them a secondary source of contamination (Bagchi, 1987).

(ii) It is mainly clay minerals that are involved in cation and anion exchange reactions (Bagchi, 1987). Broken bonds and edges, replacement of hydrogen ions with other ions, and isomorphous substitution have been identified as sources of cation exchange capacity in clay and the nature of exchangeable cations can be linked to pH (Grim, 1962). Anion exchange capacity increases as pH reduces (Bagchi, 1987).

(iii) Suspended and settleable solids can be filtered out of leachate as it passes through soil pores by being physically trapped (Bagchi, 1987).

(iv) Chemical precipitation occurs when dissolved chemical species exceed their solubility limit. The solubility limit depends on temperate, pH, solvent, ionic species and concentration, and concentrations of other dissolved substances; pH is known to be the most important factor (Minnesota Pollution Control Agency, 1978). Precipitation of trace and heavy metals in soils is closely related to pH; at neutral or higher pH they are known to form very slowly soluble or insoluble precipitates (Fuller, 1977).

(v) Adsorption is the adhesion of molecules onto clay particles; it reduces the total dissolved solids in leachate. It is a pH dependent process limited by the soil's specific surface area, once the surface reaches capacity no further leachate can be absorbed (Bagchi, 1987) and, in some cases, material attached to the soil can be desorbed increasing the total dissolved solids in the leachate (Giles, 1970). Sorption is enhanced by increased pH (Kjeldsen et al., 2002).

(vi) Dilution does not chemically alter leachate constituents or attenuate them, instead it reduces the concentration of contaminants (Bagchi, 1987). This can only happen once the leachate has reached surface or groundwater.

A secondary source of contamination can be contaminants that have leached from the waste that have been attenuated in sediments, which can be a long-term repository, or in biota (Massoudieh et al., 2010). As attenuation processes can be affected by the chemical environment, changes to the environment through anthropogenic, e.g. dredging, and natural disturbances, e.g. storms or tidal movement, can result in the release of contaminants from sediment and pore water to the water column (Christensen et al., 1994; Eggleton and Thomas, 2004). Contaminants can also be released from

undisturbed sediments through diffusion at the sediment-water interface (Fan et al., 2013).

#### 2.3.4.2 Eroding landfill sites

If waste materials, previously trapped leachate or contaminated sediments are released, e.g. through erosion of the encapsulating materials, changes to the biodegradation conditions, adsorption processes, dissolution/precipitation and redox processes may occur that remobilise contaminants due to the resulting changes in the physico-chemical environment (Eggleton and Thomas, 2004). Oxidation of organic waste will result in the production of carbon dioxide and may decrease pH, which would increase heavy metal mobility (Bozkurt et al., 1999; Flyhammar and Håkansson, 1999).

#### 2.3.4.3 Sites subjected to inundation

Emissions from landfills under normal operating conditions are well studied, but emissions from flooded landfills are not (Laner et al., 2009). Even where encapsulating materials remain intact, future increases in water entering the landfill, due to more intense rainfall or inundation resulting from climate change effects, are likely to change leachate chemistry through the same mechanisms discussed for eroded waste. The volume of leachate generated would increase as more water would percolate into the site which in turn will cause leachate leakage to increase (Bagchi, 1994). It has been estimated that metal release during fluvial flooding of landfills may increase by up to four orders of magnitude through leaching or up to six orders of magnitude if sediment is eroded (Neuhold and Nachtnebel, 2011). The highest concentrations of metals in the leachates are likely to occur within the first 30 minutes of the waste being saturated (Belevi and Baccini, 1989).

Increasing saltwater intrusion into estuaries due to climate change effects, combined with the increased likelihood of flooding, means coastal landfills are increasingly likely to be inundated with saline waters. Although the effects of saline intrusion into landfill waste sites are poorly studied (Khoury et al., 2000) there is literature relating to the effects of salinity on contaminant mobility in a broader context. The ingress of seawater into contaminated sediments has the potential to cause dissolution of minerals and increase the extractability of iron and manganese oxide fractions (Emmerson et al., 2000; Speelmans et al., 2007). The mobility of some metals increases with salinity due

to the formation of soluble complexes with $Cl^-$ and $SO_4^{2-}$ anions and competition with $Ca^{2+}$, $K^+$, $Mg^{2+}$ and $Na^+$ cations for sorption sites (Acosta et al., 2011; Chapman and Wang, 2001). In addition, although seawater is alkaline it can result in an initial decrease in pH in sediments due to the salts causing $H^+$ exchange and hydrolysis of displaced trace metals (Wong et al., 2010) and reduced pH increases the mobility of metals. Millward and Liu (2003) found metals in estuarine sediment rapidly (minutes to hours) desorbed in saline water. However, not all researchers agree that increased salinity results in increased metal mobilisation. Suh *et al.* (2003) and Schäfer *et al.* (2009) found increased salinity may decrease metal mobility or cause no significant change. In fact, there appears to be no consensus between studies of how metal mobility changes with salinity; Table 2.9 shows the findings of a selection of studies as an example. These conflicting findings may be due to several reasons: Some researchers suggest the discrepancies may relate to differences in the types of estuaries studied (Kraepiel et al., 1997; Cenci and Martin, 2004), and soil/sediment type may influence the results (Shulkin and Bogdanova, 2003; Wong et al., 2010). Metal speciation may also be a factor where increasing salinity results in increased metal mobility (Schäfer et al., 2009; Zhao et al., 2013); however, there has been little research comparing the effects of metal speciation and salinity effects on mobility in sediments (Zhao et al., 2013). Sahuquillo *et al.* (2002) found that the mobile fraction of metal contamination in soils and sediments is not proportional to the total metal content present due to differences in how they are bound within the soils/sediments, and the mobility of some metals correlate with levels of dissolved organic carbon (Kalbitz and Wennrich, 1998). Therefore, the apparent differences in leachability may also be related to variation in how the metals were bound within the materials being leached at each salinity or the organic content of the materials. In addition, the varying durations of experiments and differences in salinity levels used may contribute to the variation in findings.

**Table 2.9: Comparison of results from studies of contaminant mobility with increased salinity in order of study duration**

| Study by | Wong *et al.* (2010) | Wong *et al.* (2013) | Schäfer *et al.* (2009) | Acosta *et al.* (2011) | Zhao *et al.* (2013) | Suh *et al.* (2003) |
|---|---|---|---|---|---|---|
| **Leach test duration** | **4 hours** | **4 hours** | **24 hours** | **24 hours** | **24 hours** | **21 weeks** |
| **Salinity** | **Synthetic seawater diluted to 0, 20, 50, 80, 100%** | | **Various** (S= 0, 15, 31) | **0, 0.1, 0.2, 0.3 M NaCl** | **Various** (0, 5, 10, 15, 20, 25, 30, 35%) | **0.05 M NaCl** |
| **Does the contaminant's mobility increase or decrease with increased salinity?** | | | | | | |
| **Al** | increases | decreases | | | | decreases |
| **Cd** | | | increases | increases | increases | |
| **Co** | | | increases | | | |
| **Cr** | | | | | | decreases |
| **Cu** | | increases | increases | increases | increases | decreases |
| **Fe** | increases | increases | | | | decreases |
| **Mn** | | increases | | | increases | increases |
| **Mo** | | | unchanged or decreases | | | |
| **Ni** | increases | increases | increases | | | |
| **Pb** | | | unchanged or decreases | increases | increases | decreases |
| **V** | | | unchanged or decreases | | | |
| **Zn** | increases | | increases | unchanged | | decreases |

It is known that persistent organic pollutants (POPs) generally have low aqueous solubility, quickly sorbing to inorganic and organic particles in water (Neff, 1979; Carron, 2003; Fadum, 2009; Tsibart and Gennadiev, 2013). In contrast to metals, the effects of salinity on organic compound mobility are relatively well understood and it is known that, except in some rare cases, organic compounds are less soluble in saline waters than freshwaters due to the 'salting-out' effect (Xie et al., 1997; Kim and Osako, 2003). The exceptions to this are some organic contaminants that carry a charge, e.g. tributyltin, which exhibit the opposite behaviour if the pH is 6 or lower (Burton et al., 2004). Where contact times have been many years in historically contaminated sites, organic contaminants sorbed to particles can be very slow to desorb when inundated, i.e. days to months (Pignatello and Xing, 1996). Therefore it is likely that organic

contaminant release would not increase if the waste were exposed to saline waters and may in fact decrease.

In summary, saline intrusion into historic coastal landfill sites is unlikely to increase organic contaminant concentrations in leachate, but it is currently unclear how it may affect metal mobility due to the very limited research that has been carried out regarding salinity effects on landfill waste and the conflicting findings of broader research into salinity effects on inorganic contaminant mobility. If metals are more mobile in saline conditions then exposure of the waste to saline waters may have a significant environmental effect as it has been estimated that 99.9% of metals deposited in landfill waste are retained after about 10 years (mean waste age) (Belevi and Baccini, 1990) suggesting that there is potential for a high levels of metals to be released from historic landfills if the chemical environment changes. Therefore, there is a clear need for a study to be undertaken analysing the solubility of metals from historic landfill waste in fresh and saline water.

### 2.3.5 Pollution from coastal landfills - receptors and ecotoxicology

There have been limited studies of the effects of leachates on estuarine and marine environments and none on the effects of landfill debris (Pope et al., 2011; Njue, 2012). Whether historic coastal landfills cause pollution depends upon whether there are any pathways for contaminants they contain to reach receptors that may be adversely affected (Environment Agency, 2003b). Many historic coastal landfills are located in or within close proximity to sensitive receptors (see Table 2.10 and Table 2.11). Land-based flora can be affected by contaminants through uptake via their roots and can bio-accumulate contaminants such as heavy metals (Kvesitadze et al., 2009), which can result in trophic transfer of contaminants through the food web. Contaminants carried by leachate can enter ground and surface waters, which may be a pathway to consumption by humans via abstraction for drinking water or irrigation purposes. These contaminants may also have adverse effects upon a variety of other flora and fauna that utilise coastal environments, e.g. through direct toxicity, eutrophication, deoxygenation or biomagnification/bioaccumulation leading to toxicity (Pope et al., 1999; Njue, 2012). Erosion of historic coastal landfill sites can release contaminated materials that can become incorporated into sediments (Pope et al., 2011). Eroded waste materials that float, e.g. wood and plastics, may provide a pathway for contaminants from landfill sites

to reach even the remotest of locations around the globe (Barnes et al., 2010). Plastics can carry high concentrations of inorganic and organic contaminants which increase the longer the plastics have been exposed to the contaminants and can exceed concentrations on natural sediments (Ashton et al., 2010; Rochman et al., 2014).

**Table 2.10: Number of historic coastal landfills in or within 100 m of sensitive environmental areas**

| Site type | Number of landfills in or within 100 m of sensitive sites | Proportion of the 1264 historic coastal landfills (%) |
|---|---|---|
| SSSI | 414 | 33 |
| National Nature Reserve | 33 | 3 |
| SAC | 171 | 14 |
| SPA | 299 | 24 |
| Ramsar | 292 | 23 |
| Marine Protected Areas[1] | 245 | 19 |
| Bathing Water Catchments | 574 | 45 |
| Bivalve mollusc production areas[2] | 46 | 4 |

[1]OSPAR MPAs
[2]Some locations have multiple bivalve production areas designated for different species
Source: Analysis of Natural England (2016), JNCC (2016), Environment Agency (2013a; 2015b) and Cefas (O. Morgan, pers. comm., email, 2/11/2015) GIS datasets in ESRI ArcMap.

**Table 2.11: Number of sensitive environmental areas on or within 100 m of historic coastal landfills**

| Site type | Number on or within 100 m of coastal landfills |
|---|---|
| SSSI | 121 |
| National Nature Reserve | 20 |
| SAC | 28 |
| SPA | 39 |
| Ramsar | 36 |
| Marine Protected Areas[1] | 43 |
| Bathing Water Catchments | 128 |
| Bivalve mollusc production areas[2] | 137 |

[1]OSPAR MPAs
[2]Some locations have multiple bivalve production areas designated for different species
Source: Analysis of Natural England (2016), JNCC (2016), Environment Agency (2013a; 2015b) and Cefas (O. Morgan, pers. comm., email, 2/11/2015) GIS datasets in ESRI ArcMap.

The Environment Agency (2011) has published clearly defined acceptable limits of chemical concentrations in water and for discharges of liquids to water (Environment

Agency, 2013c); these are intended to minimise risk to humans and aquatic life. It is important to note that the standards only consider the effects of each contaminant in isolation, they do not consider the effects of combinations of contaminants, and effects of exposure to complex chemical mixtures such as those found in leachates are not well documented (Melnyk et al., 2014). Consequently there have been a number of studies relating to toxicity of leachate and proposals have been made for using bioassays for assessing pollution from leachate, but none have been implemented to date (e.g. Isidori et al., 2003; Melnyk et al., 2014; Linderoth, 2015).

As discussed previously, contaminants may be attenuated in sediments; however, there is limited and conflicting research on whether the input of metals from coastal landfills is significant in estuarine sediments. Some researchers suggest the input of metals from coastal landfills is significantly less than from other sources (particularly sewage) and ratios of metals to total organic carbon increase close to landfills, but metal concentrations in sediment do not (e.g. Seidemann, 1991). Other researchers suggest that concentrations of metals do increase near landfills (Denton et al., 2009; O'Shea, 2016) and bioaccumulation may result in harmful concentrations of heavy metals in sediments, flora and fauna (Pope et al., 2011; Njue, 2012). The discrepancy in findings may be related to continual erosion of sediments in some environments meaning there is little evidence remaining around the landfill sites to show that leaching has occurred (Denton et al., 2007; Denton and Morrison, 2009).

Heavy metals are toxic, non-biodegradable and accumulative, making them of particular concern in aquatic environments (Fisher and Reinfelder, 1995; Lau, 2000); contaminants in sediments can have adverse effects on aquatic life such as reduced growth and fecundity, developmental abnormalities or increased mortality (Eggleton and Thomas, 2004; Matthiessen and Law, 2002; Hübner et al., 2009). It is currently unclear at what concentrations contaminants in sediment are considered to be detrimental to flora and fauna in UK waters as there are no quantified UK Environmental Quality Standards (EQSs) for *in situ* sediment (Spencer and MacLeod, 2002; Port of London Authority, n.d.). The lack of EQSs is related to the difficulty in defining pass/fail limits on contaminants that are appropriate to both suspended and deposited sediments, that take into account bioavailability, and are appropriate for various organisms and areas (Crane, 2003; Hübner et al., 2009). In the absence of UK

sediment EQSs, the UK Marine SAC Project (Cole et al., 1999) recommends using the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life (Canadian Council of Ministers of the Environment, 2002) for initial assessment of toxic substance concentrations in sediment. For contaminants not included in the Canadian guidelines, the NOAA Screening Quick References Tables (Buchman, 2008) may also be a useful reference. As many of the landfills under consideration are located on estuaries, these guidelines, which were developed for marine and freshwater systems, may not be entirely appropriate; however, more suitable guidelines or standards are not currently available (Chapman and Wang, 2001).

It is also worth noting that the materials within historic landfill sites could be ecologically harmful if released to the surrounding environment regardless of their contaminant concentrations. Plastics in particular are known to have detrimental effects on marine fauna as they can be inhaled, ingested and cause entanglement of species of all sizes, from small birds to whales (Derraik, 2002; Teuten et al., 2009). The effects are typically mechanical, e.g. gastrointestinal blockages, and are often fatal, but can also be a result of the contaminants the plastic carry (Derraik, 2002).

## 2.4 Ranking the pollution risk posed by landfill sites

The very large number of historic coastal landfill sites around England means it will not be possible to protect all of them indefinitely from climate change effects as there are limited coastal management resources. Coastal defence budgets are already stretched and central government funding is continuing to be reduced whilst flood defence maintenance costs are rising £10-£30 million plus inflation per year (Bennett and Nartwell-Naguib, 2014). If analysis of waste from historic coastal landfills shows that erosion of the contaminated materials or leaching of metals from them would breach environmental quality standards/guidelines, it will be essential to know which historic coastal landfills present the greatest environmental risk in order to prioritise the sites for remedial actions (Kumar and Alappat, 2005; Sharma et al., 2008; Singh et al., 2009). Also, it is unlikely that funding would be made available without demonstrating the level of environmental risk (Alaska Department of Environmental Conservation, 2015).

Risk is typically considered to be a function of probability and consequence (Wamsley, 2015). Therefore, to assess the pollution risk posed by a coastal landfill site it is necessary to understand the probability of its waste leaching harmful levels of contaminants and/or waste being eroded and released from the landfill site, and to understand the consequences of those events happening (Neuhold and Nachtnebel, 2011). There are many factors that may influence the probability of a site leaching contaminants or releasing waste, including the condition of any flood defences, the design standard of those defences and the probability of the design standard being exceeded, the area of inundation, the mechanical properties of the waste, e.g. waste cohesion, whether the coast is eroding at the landfill's location, site exposure, whether the site is vegetated or armoured and the presence of any buffers, e.g. saltmarshes, site maintenance regime, climate change related sea level rise and storm severity (Cooper et al., 2013; Alaska Department of Environmental Conservation, 2015). The consequences of waste or leachate entering the surrounding environment will depend upon the quantity of leachate or waste released and its contaminant concentrations, contaminant mobility, dilution by the receiving waters, and receptor sensitivity (Laner et al., 2008b; Sharma et al., 2008). The quantity of waste or leachate released will depend on many of the same factors as the probability of release, plus the size of the landfill, whether it is divided into structurally stable cells, the shape of the landfill, e.g. the proportion of it

adjacent to the coast, elevation of different areas, and how quickly any breach can be repaired (Stanczak and Oumeraci, 2012; Liang et al., 2015). Many of these data are not readily available and would require impracticable levels of resources to obtain for all of the 1264 landfill sites around the coast and estuaries of England, which means it is not possible to investigate all landfills in detail. Where detailed data are not readily available to assess risk at local, regional or national scales Rosendahl Appelquist and Balstrøm (2014) propose a three step approach to assessment, where steps 1 and 2 are used for regional or national scale assessments and step 3 is only used for local scale assessments:

Step 1. High level initial screening using remote sensing and existing data to gain a cost-efficient, relatively low accuracy overview of the risk.
Step 2. Field verification of the data used in step 1.
Step 3. Systematic and detailed field investigations for high accuracy, local level assessments of risk hot-spots identified in steps 1 and 2.

Accordingly, a screening level assessment using existing or easily obtained data to provide an indication of the risk of pollution from historic coastal landfills, and support prioritising which sites should be investigated in detail, would be a valuable tool for agencies with coastal management responsibilities, e.g. the Environment Agency and local authorities (Kumar and Alappat, 2005; Sharma et al., 2008).

There are a variety of methods used to interpret complex data to assess coastal vulnerability to climate change. The European Environment Agency groups the commonly used methods into four main categories: index-based methods, indictor-based methods, GIS-based decision support systems, and dynamic computer models. These methods can provide a simple numerical ranking to understand where risks may be relatively high (Ramieri et al., 2011).

Coastal vulnerability indices are management tools used to simplify diverse data types, representing complex and interacting parameters, into a readily understood form that indicates their combined effect. They are usually undertaken at global or regional scales and each one is tailored for a particular management need based on which data are available and appropriate to assess vulnerability, and the time required to process them (McLaughlin and Cooper, 2010). A coastal vulnerability index usually takes into

account 6 or 7 key parameters and reports the vulnerability as a single, often unitless, number that does not explain the assumptions and aggregations that led to it unless the calculation method is also given (Ramieri et al., 2011).

In contrast, vulnerability indictors express their findings as a serious of independent factors e.g. pressures, sensitivity and damage, each represented by a groups of indicators (parameters). These may also be combined into a final summary indicator and GIS tools may be used to facilitate this. This approach makes it easier to understand the critical factors in the published results (Ramieri et al., 2011).

GIS-based decision support systems (DSSs) use vulnerability indices and indicators combined with validated numerical models of climate change scenarios to generate a variety of hazard, risk and damage maps. Dynamic computer models build upon these to incorporate changes with time (Ramieri et al., 2011; Musekiwa et al., 2015).

There have been a number of attempts to apply the index and indictor approaches to the management of landfill sites both on the coast and inland, none of which have been widely adopted. These are detailed in the following sections.

### 2.4.1 Leachate pollution index (LPI)

The Leachate Pollution Index (LPI) was developed by Kumar and Alappat (2005) to rank landfill sites for their leachates' pollution potential in order to support resource allocation for remediation works. They surveyed eighty experts to identify and rank the eighteen most important leachate contaminants and rate their pollution potential at different concentrations. They then developed a weighted formula for calculating the LPI based on comparing concentrations of those contaminants in leachate samples to the ratings. The LPI provides a snapshot assessment that comparatively rates sites, but does not consider the volume of leachate each site produces, whether the leachate has a pathway to receptors or receptor vulnerability.

### 2.4.2 Environment-Landfill Interaction Index (ELI)

This method developed by Calvo *et al.* (2007) considers more pollutant types than the LPI method (above), e.g. gases, and it takes into account pollutant pathways and the environmental value of receptors. It focusses on currently operating landfills and assigns an Environmental Risk Index (ERI) based on the probability a receptor will be

contaminated and the receptor's environmental value. Each receptor is assigned an Environmental Weighting Coefficient (EWC), and an overall Environment-Landfill Interaction Index or Impact Index (ELI) is calculated by multiplying the ERI and the EWC and summing the results for all receptors. Although it is noted that site visits were necessary to assess the 17 landfills that were ranked, no details are provided as to how the probability of contamination, the receptor's environmental value or the EWC have been determined, making it difficult to apply this method elsewhere.

### 2.4.3 Landfill pollution potential index for uncontrolled landfills

Sharma *et al.* (2008) developed the Landfill Pollution Potential Index (LPPI) using similar development methods to the LPI and ELI. Experts were surveyed to determine six environmental parameters to include in assessments and to weight their importance. In weighted order (high to low) the parameters selected were: sub-surface water, surface water, air, aesthetics, flora and fauna, and noise. A severity factor is determined for each parameter primarily based on existing environmental assessment methods: the LPI for sub-surface water quality (Kumar and Alappat, 2005), the National Sanitation Foundation Water Quality Index for surface water quality (Brown et al., 1970), gas measurements for air quality, occupational safety limits for noise, the type of ecosystem around the landfill for flora and fauna, and the Landfill Aesthetics Index for aesthetics. This method improves upon the ELI by providing comparatively clear methods for scoring the severity of the effects of each parameter.

The LPPI only assesses current pollution not the potential for future pollution from a site and it does not consider the possibility of sites eroding, therefore it cannot be directly applied to determining the potential for pollution from historic coastal landfills.

### 2.4.4 Groundwater contamination hazard rating for landfills

Singh *et al*. (2009) developed a system for rating the potential for groundwater contamination from landfills using similar development methods to the LPI, ELI and LPPI. Existing methods were reviewed to determine which parameters they utilised, and a panel of experts was surveyed to determine how they would rate the importance of the parameters when considering groundwater pollution from landfills. The ratings were then used to determine weightings that the parameters should be assigned. Three main categories of parameter were identified: source, pathway and receptor. Hazard ratings

for each category were calculated before being aggregated into an overall hazard rating. The source hazard rating includes the quantity of waste, waste composition and throughput of water. The pathway hazard rating includes assessment of the leachate containment systems, vadose zone thickness and permeability, and leachate movement through the aquifer and distance to groundwater extraction points. The receptor hazard rating only considers groundwater use.

The groundwater contamination hazard rating method only assesses groundwater pollution, it does not consider surface water pollution or the possibility of sites eroding; therefore, it cannot be directly applied to determining the potential for pollution from historic coastal landfills.

### 2.4.5 Prioritisation of landfills subject to flood risk in Serbia

Okaneya *et al.* (2013) proposed a simple preliminary screening method to determine which landfill sites are at the highest risk of fluvial flooding where data are limited. The method only considers whether a landfill is over a certain size, within 700 metres of a river greater than 50 m in width, whether the landfill is elevated above the floodplain and whether there are residential areas downstream. The site's waste and leachate composition are not considered, nor is the potential for pollution of other receptors or the cumulative effect of multiple small landfills within a reach. In addition, contaminant mobilisation and the potential for waste to erode are not taken into account.

### 2.4.6 Austrian fluvial flooding of landfills risk assessment

Laner *et al.* (2008b; 2009) and Neuhold *et al.* (2011; 2013) have developed a more detailed risk assessment of the pollution risk from fluvially flooded landfills, which considers contaminants leaching from sites during flooding and the potential for sites to erode releasing waste, but only assesses subsequent contamination of the water column not sediment against environmental quality standards. Laner *et al.* (2008b) use two contaminant emission scenarios, the first where the landfill maintains structural integrity and the second where some of it erodes or collapses. Neuhold *et al.* (2011) expand these into four scenarios; I (low), II (medium) and III (high) relate to soluble contaminant movement through flooding and scenario IV refers to loss of waste through erosion or catastrophic failure caused by changes in water content (Blight and Fourie, 2005). It is unclear exactly what scenarios I-III represent or how they have been individually

assessed as they are presented as a single result, but it is probable they have been developed from a previous technical report by (Laner et al., 2008a) to represent the following:

I – Low: Largely homogeneous flow of water through the landfill, minimum potential mobilisation of contaminants during flooding.

II – Medium: Average homogeneous flow of water through the landfill, average potential mobilisation of contaminants during flooding.

III – High: Diverse flow of water through the landfill, maximum potential mobilisation of contaminants during flooding.

Laner *et al.* (2008b; 2009) approximate contaminant (substance) loads released in the event of landfill waste being eroded using Equation 2.1 and Equation 2.2, and approximate the contaminant load released if a site is flooded, but not eroded using Equation 2.3. The assumption is made for the erosion calculations that the entire body of waste is released. It is also assumed that all landfill sites contain the same contaminant concentrations and that all sites release the same proportions of their contaminants when inundated. In Equation 2.3 the proportion of the waste participating in the water flow is assumed to be 70%, an assumption Laner *et al.* (2008b) made based on research by others into preferential flow paths in landfills.

**Equation 2.1: Total mass of substance (i) released through landfill erosion ($S_i^A$)**

$$S_i^A = m_w \; x \; c_i \; x \; R_i$$

*Where:*
*$m_w$ = (eroded) mass of waste (kg)*
*$c_i$ = initial content of substance (i) in the waste at the time of landfilling (kg kg$^{-1}$), from Belevi and Baccini (1989)*
*$R_i$ = Remaining fraction of the initial content of the substance at the time of erosion, from Baccini et al. (1987) and Döberl (2002)*

**Equation 2.2: Soluble mass of substance (i) released through landfill erosion ($S^S_{mi}$)**

$$S^S_{mi} = S_i \; x \; c_{mi}$$

*Where:*
*$S_i = S_i^A$ (see Equation 2.1)*
*$c_{mi}$ = soluble content of substance as a proportion of the total amount, from Belevi and Baccini (1989)*

**Equation 2.3: Total mass of substance (i) released through landfill flooding ($S_i^B$)**

$$S^B_i = m_w \; x \; q_i \; x \; F$$

*Where:*
*$m_w$ = (flooded) mass of waste (kg)*
*$q_i$ = soluble content of the substance (kg kg$^{-1}$), from Belevi and Baccini (1989)*
*F = Fraction of the waste volume participating in water flow*

To determine the risk of pollution, Neuhold and Nachtnebel (2011) considered whether the contaminant load predicted using these equations would breach Austrian Landfill Directive leaching limits once dilution by flood waters is taken into account. They then developed the Flood Risk Evaluation Matrix (FREM) shown in Figure 2.6. The area of the landfill inundated, the area likely to erode (based on flow velocities and shear stress from water movement) and the pollution risk are all considered in order to qualitatively determine the pollution risk from 1 in 30, 1 in 100 and 1 in 300 fluvial flooding events. Site specific modelling was required to obtain these parameters; consequently, the method cannot be readily applied to prioritise large numbers of sites, and so far it has only been applied to a limited number of case studies which were selected by screening for sites at risk of flooding and then modelling them in detail (Neuhold, 2013).

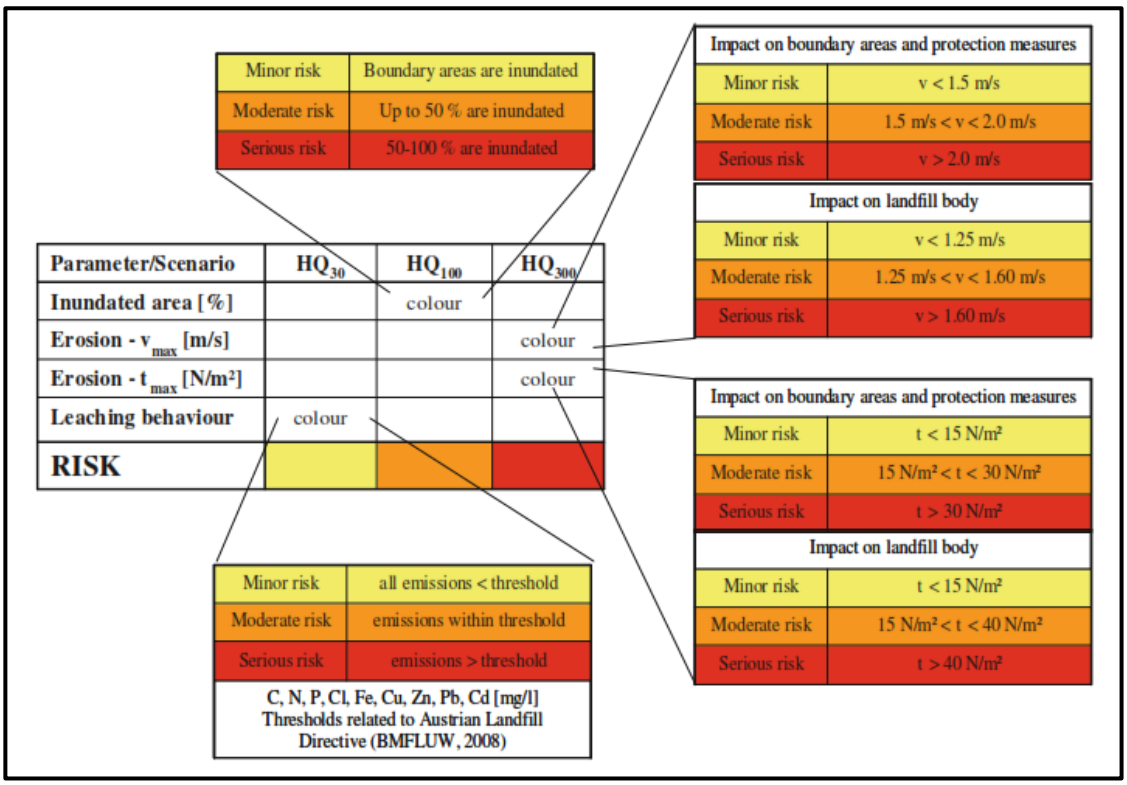



**Figure 2.6: Flood Risk Evaluation Matrix, input parameters and thresholds (Neuhold and Nachtnebel, 2011, p. 365)**

This method cannot be directly applied to the coastal landfills of interest in this research due to differences in the flood scenarios used in the UK and the current lack of data relating to contaminant solubility in saline waters. In addition, available UK coastal flood models do not consider shear stress from water movement (A. Barber, Environment Agency, pers. comm., email, 2/1/2014), which was used by this method to assess erosion; so, a significant amount of new modelling would be required when considering the risk posed by 1264 coastal landfills around the coast of England, which would be restricted by available resources.

### 2.4.7 The WEAR project

The Waste Erosion Assessment and Review (WEAR) project was a comprehensive four year study by the Alaska Department of Environmental Conservation which assessed 716 sites for their potential to pollute (Alaska Department of Environmental Conservation, 2015). The study assessed all potentially contaminated sites that were considered to be at risk of eroding within 50 years, e.g. landfills, fuel tanks and industrial sites. Site location, proximity to receptors, probable contaminants, erosion type and potential for environmental or human exposure were used to score each site for

erosion and contaminant risk. The scores were based on the likelihood of erosion within 50 years (sites likely to erode within 5 years were rated as the highest risk) and the likelihood of erosion releasing contaminants. The scores included weightings reflecting the importance of different measure parameters, e.g. estimated time to erosion was weighted above the causes of erosion. No samples were taken from sites, contaminants likely to be present were assumed based on the site type and professional judgement. Whether erosion mitigation has been attempted is taken into account in the scoring, but not whether the mitigation is effective. The potential for contaminants to leach was not considered. The resulting risk levels assigned to contaminated sites are comparative not absolute, with the highest risk sites considered to be those that are within the top 25% for erosion risk and top 25% for contaminant risk.

### 2.4.8 CIRIA C718 Source-Pathway-Receptor assessments

CIRIA C718 'Guidance on the management of landfill sites and land contamination on eroding or low-lying coastlines' contains guidance on the use of Source-Pathway-Receptor conceptual models and the use of probability vs. consequence charts to determine risk severity for individual landfill sites in the UK. Although it is not an index or indictor method and it does not rank the risk from multiple landfill sites, CIRIA C718 has been included in this review as it is currently the standard method for assessing individual landfill sites in the UK.

The guidance advises that the landfill waste (the source) should be classified as inert, non-hazardous or hazardous so that potential alternative disposal sites can be determined as part of the assessment. However, most historic landfill sites precede the introduction of these landfill categories and, therefore, Environment Agency records do not contain the necessary data to classify waste within existing sites in this way (Environment Agency, 2015b). Therefore, to use this method for ranking sites would require sampling and analysis of waste from all 1264 historic coastal landfills, which would be impracticable due to the costs involved. The guidance includes a number of landfill waste release scenarios (the pathways) that should be assessed, but does not consider leaching of contaminants from sites where the capping materials are intact or define how to assess the probability of the scenarios occurring, although it does offer some general advice on the types of datasets to use. The guidance also defines which receptors should be considered to determine the consequences of waste being eroded,

but does not define how to categorise the severity of the consequences. The data used to assess the source, pathways and receptors are left to the assessor's discretion, as is the choice of probability-consequence matrix used to categorise the overall risk, so assessments carried out by different coastal managers are unlikely to be comparable, which would be problematic when attempting to rank sites to allocate resources for landfill remediation on regional or national scales.

### 2.4.9 Summary of landfill ranking methodologies

Each of these landfill pollution risk ranking methods have been developed for use in a specific environment, either by utilising available or easily obtainable data where large numbers of sites are being assessed, or by greatly restricting the numbers of sites ranked where data are difficult or expensive to obtain or site specific modelling is required. These methods provide useful information regarding which factors are important to consider when screening landfills for pollution risk. However, as no single method considers pollution from eroded materials and leaching, and screens large numbers of sites without extensive long-term studies, none of the methods provides a solution to identifying which of England's historic coastal landfill sites are the highest priority for further investigation and remediation. Therefore, there is a clear need for a screening method to be developed that can utilise existing or easily obtained data to rank which historic coastal landfills present the greatest environmental risk in order to be able to prioritise the sites for further investigation and remedial actions.

## 2.5 Summary

Historic coastal landfills may have the potential to pollute the aquatic environment and the risk of them doing so is likely to increase due to climate change effects, e.g. increased coastal erosion and more frequent coastal flooding. The contents of these landfills are poorly documented and there is little published research into the concentrations of contaminants in the solid waste historic landfills contain. In addition, although there exists a large body of literature relating to contaminant mobilisation from waste through freshwater ingress, little has been published on the effects of saline waters on contaminant mobility from landfill waste and, therefore, what effect increased saline intrusion into estuaries and saline flooding may have on these sites. Although under normal operating conditions, i.e. when sites are not flooded and the capping materials are intact, Ammonium-N is considered to pose the greatest pollution risk in leachates with metal concentrations being negligible, it is likely that saline waters will mobilise the metals that are present in the solid waste, but no previous research has investigated this.

This lack of understanding of historic landfills means that site managers cannot adequately assess the risk associated with non-conservative coastal management options such as managed realignment or no active intervention and, consequently, hold the line is the default management option for historic coastal landfills. Therefore, understanding the contaminant concentrations in the waste, the response of metals to salinity changes and whether contaminants would be released at a level that would cause pollution in the event of a site being eroded or catastrophically breached, could inform the future management of coastal and estuarine landfill sites at a local scale. A method for assessing and ranking the pollution risk of historic coastal landfills would enable coastal managers to prioritise resources for further investigation, maintenance or improvement of these sites where required at regional and nation scales.

# Chapter 3. Contamination of the intertidal zone by solid waste materials eroded from historic coastal landfills

## *Introduction*

## *3.1 Aims and objectives*

Historic coastal landfills have the potential to pollute the aquatic environment and the risk of them doing so is likely to increase due to climate change effects, e.g. increased coastal erosion and more frequent coastal flooding. There has been little previous research into contaminant concentrations in solid waste as leachate movement is considered to be the primary pathway for contaminant release from waste when landfill sites are undisturbed (e.g. Robinson and Maris, 1979; Robinson et al., 1982; LaGrega et al., 1994; Robinson, 1995; 2007; Ziyang et al., 2009). Only one published study (Prechthai et al., 2008) measured inorganic contaminants in solid waste from a historic landfill site using methods that allow comparison to sediment quality guidelines. Two others used hydrofluoric acid extractions (Zhao et al., 2007; Quaghebeur et al., 2013), which are not suitable for determining metal concentrations for comparison to sediment quality guidelines (Canadian Council of Ministers of the Environment, 2001), and one did not reported the extraction method used (Hull et al., 2005).

There exists some grey literature where contaminant concentrations in the basic matrix from historic landfill sites, i.e. the fine and medium grained soil-like particulate materials in the solid waste (Gharabaghi et al., 2008), hereon referred to as matrix material, have been measured using aqua regia extractions (e.g. Halcrow Group Ltd, 2012), which are suitable for determining metal concentrations for comparison to sediment quality guidelines (Canadian Council of Ministers of the Environment, 2001), but other waste components are not considered. This is because British Standards for investigating contaminated land, and contaminated land and landfill investigation guidance documents are primarily concerned with assessment of contaminants in soil, water and gases, and do not require assessment of other materials (e.g. Rudland et al., 2001; British Standards Institution (BSI), 2011; Cooper et al., 2013). Therefore, it is unknown whether solid waste released into the intertidal zone as a result of historic landfill sites eroding would contain inorganic or organic contaminants at concentrations

that could cause pollution. It is also unknown whether materials other than the matrix material, e.g. wood, paper and textiles, in historic landfills contain significant concentrations of contaminants, which may have been present when the materials were landfilled or could have been attenuated by them as leachate has moved through the waste (Bagchi, 1987).

In addition, existing landfill risk assessments that attempt to rate the pollution risk from eroding waste have used two approaches to account for contaminant concentrations when comparing the relative risk of different landfill sites: In the Austrian landfill risk assessments it has been assumed that all sites contain the same contaminant concentrations, which have been used with assumptions on leaching behaviour to determine whether eroded waste would result in breaches of water quality standards (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013). In the Alaskan Waste Erosion Assessment and Review method the risks from different sites have been ranked depending on the types of materials landfilled within them, but no attempt has been made to compare contaminant levels to environmental quality standards (Alaska Department of Environmental Conservation, 2015). Therefore, understanding whether there are significant differences in contaminant concentrations between landfill sites is necessary to determine whether either of these approaches is appropriate when assessing the risk of pollution caused by waste eroding from historic coastal landfills.

The aim of this chapter is to determine the potential for contamination of the intertidal zone if solid waste materials are eroded from historic coastal landfills. This has been investigated by addressing the following objectives, to:

1. Determine the size of the waste cells in a waste filled flood embankment in order to determine the likely maximum extent of any breaches and, therefore, the maximum volume of waste likely to be released.
2. Characterise the magnitude and variability of inorganic and organic contaminant concentrations in landfill sites with different waste mixtures to determine whether contaminant data is representative of landfill sites in general and could therefore be used as generic data for regional or national scale risk assessments. Consider:
    a. Local variability of contaminant concentrations in different materials.

b. Differences in contaminant concentrations in individual material types from different areas of the same landfill site.
c. Differences in contaminant concentrations between sites.

3. Determine the potential for pollution if the waste materials erode and are released into the intertidal zone by comparing contaminant concentrations in different materials to sediment quality guidelines.

## 3.2 Research sites

Two landfill sites were chosen for waste sampling, Hadleigh Marsh, which is a flood embankment (Plate 3.1), and Leigh Marshes, which is a recreational area (Plate 3.2). Both sites are located on the Thames Estuary in Essex between Benfleet and Leigh-on-Sea railway stations (Figure 3.1).

**Plate 3.1: Hadleigh Marsh flood embankment (looking east) (J. Brand, 29/2/2014)**

**Plate 3.2: Leigh Marshes recreational area (looking east) (J. Brand, 16/3/2014)**

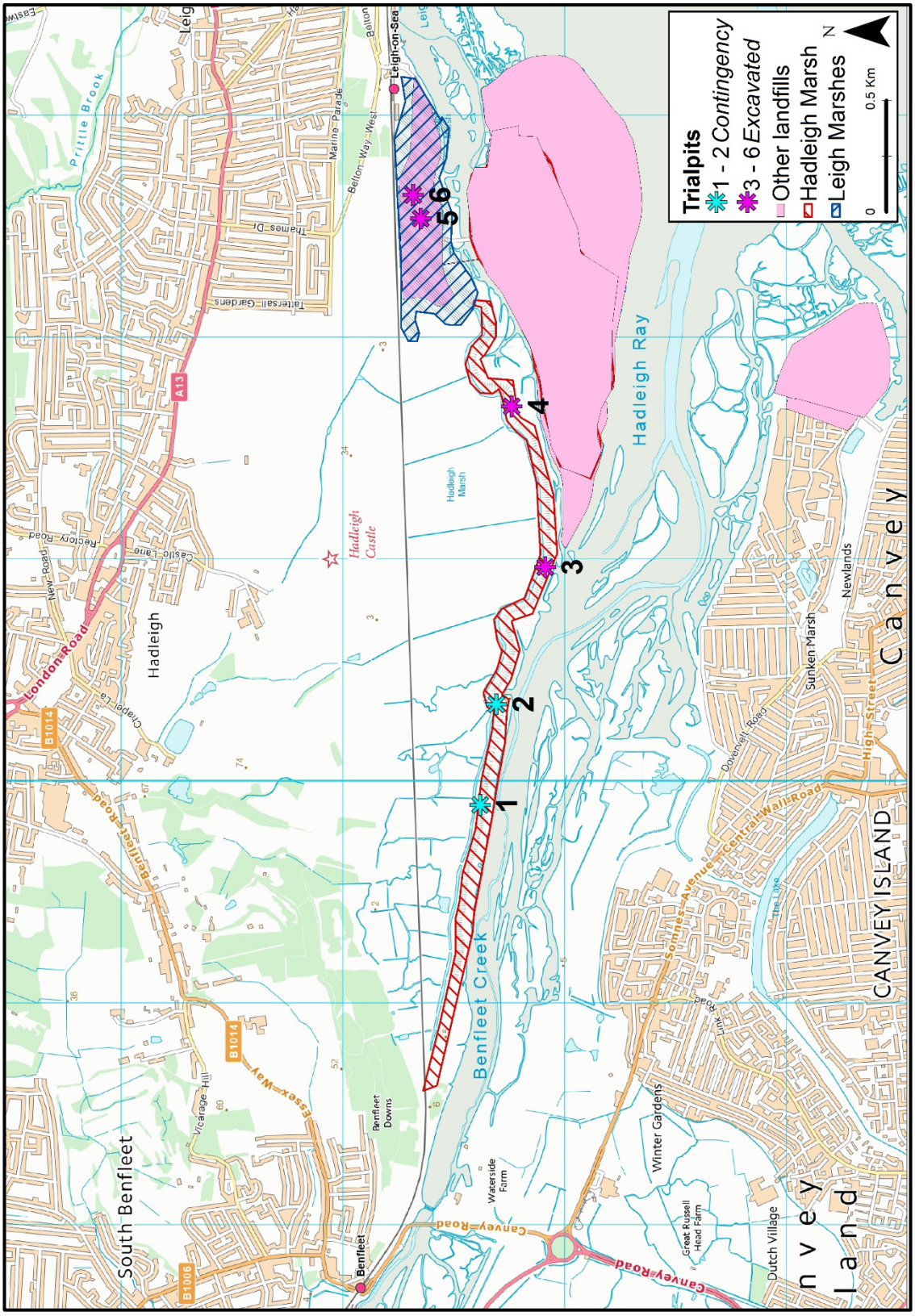



**Figure 3.1: Hadleigh Marsh flood embankment, Leigh Marshes recreational area, and the trial pit locations (Contains Ordnance Survey data © Crown copyright and database right 2014)**

### 3.2.1 Site descriptions

Hadleigh Marsh landfill dates from the 1980s (constructed between 1980 and 1987) and was selected as it is one of three coastal flood embankments in Essex constructed using household and commercial waste capped with puddled clay. The other two waste filled embankments are located at Dengie and South Fambridge, the latter also contains industrial waste. See Table 2.2 for details of landfill site waste classifications. Figure 3.2 shows the typical constituents of household waste at the time Hadleigh Marsh landfill was constructed, other waste streams are not well studied and no equivalent data could be found for the typical constituents of commercial or industrial sites (Burnley, 2007a; Parfitt and Bridgwater, 2010). A typical cross-section of Hadleigh Marsh landfill is shown in Figure 3.3. Hadleigh Marsh landfill is of particular interest, as it has a high risk of failure due to its exceptionally weak foundations (Environment Agency, 1996) and has previously been identified for managed realignment, although a hold the line management policy is now in place due to the presence of landfill waste (Environment Agency, 2009b). The landfill is approximately 4 km long, 65 m wide, 0.26 $km^2$ in area, and is estimated to contain 500,000 $m^3$ of waste (Essex County Council, n.d.), it has a minimum crest height of 4.96 mAOD (1 in 100 to 1 in 200 standard of defence) and a typical crest height of 5.8-6 mAOD (1 in 1000 to 1 in 5000 standard of defence) (Halcrow Group Ltd, 2012). Records indicate the landfill was constructed of seven waste cells (Environment Agency, 1996), no data have been found to indicate the size of the cells.

**Table 3.1: Classifications of waste in Leigh Marshes and Hadleigh Marsh landfills**

| |
|---|
| **Industrial (Leigh Marshes only):**<br>Waste from a factory or industrial process. It excludes waste from mines, quarries and agricultural wastes. |
| **Commercial:**<br>Waste from premises used wholly or mainly for trade, business, sport, recreation or entertainment. Excludes household and industrial waste. |
| **Household:**<br>Waste from dwellings of various types including houses, caravans, houseboats, campsites, prisons and wastes from schools, colleges and universities. |
| Adapted from: Environment Agency (2013f) |

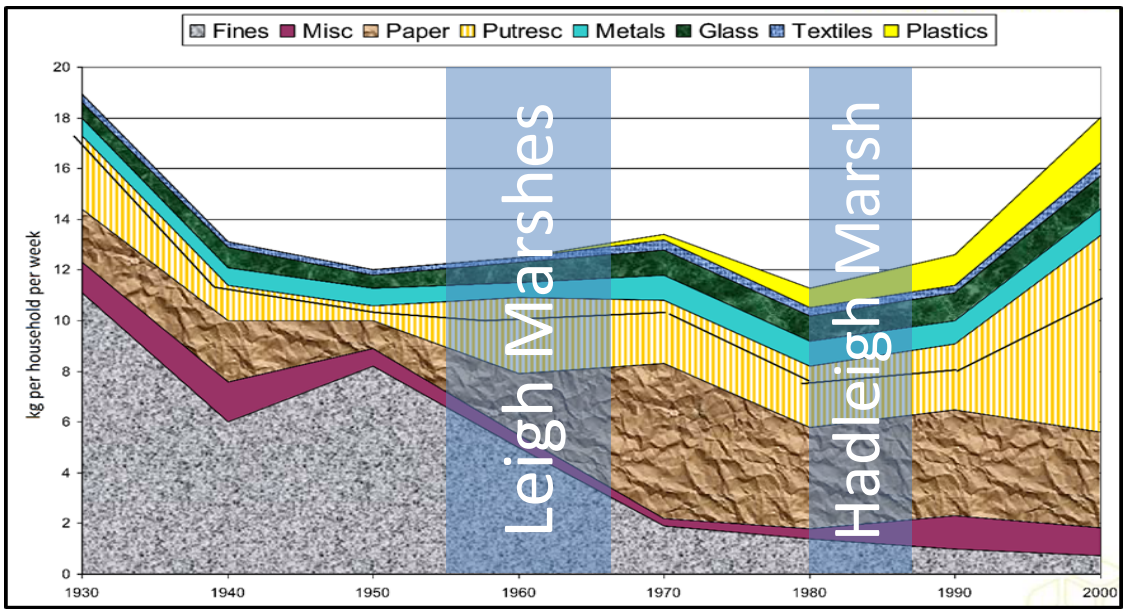



**Figure 3.2: Trends in dustbin waste in England from 1930 to 2000 (Parfitt, 2009, p. 3. Reproduced with the kind permission of Resource Futures) compared to the construction dates of the study sites**

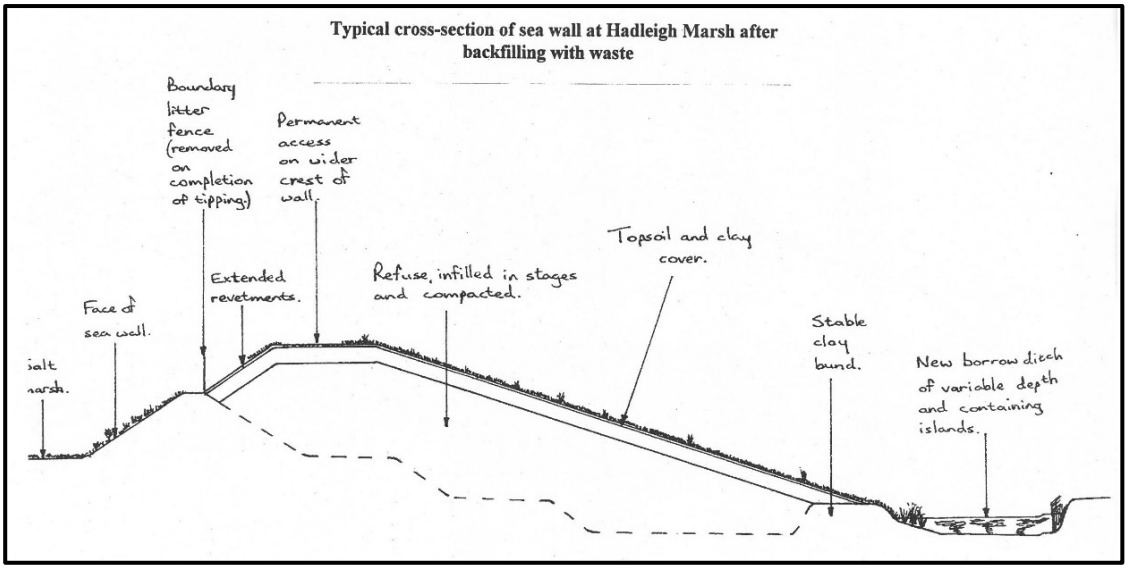



**Figure 3.3: Typical cross-section of Hadleigh Marsh waste filled embankment (Environment Agency, 1996, Appendix A7.10b)**

Leigh Marshes landfill was selected as it represents landfill sites protected by flood embankments and contains a different combination of waste streams to Hadleigh Marsh landfill, specifically industrial, commercial and household, and from a different era (1955-1967) (Environment Agency, 2015b). Figure 3.2 shows the typical constituents of household waste at the time the landfill was constructed. Leigh Marshes landfill is approximately 1.2 km by 350 m maximum and 0.25 km$^2$ in area (Environment Agency,

2015b). No data exist on the volume of waste contained within Leigh Marshes landfill. Site investigations by Halcrow (2012) included 15 trial pits to a maximum depth of 3.2 m, only two reached the natural substrate. This suggests there may be in excess of 800,000 $m^3$ of waste in the site.

### 3.2.2 Current site use

Hadleigh Marsh and Leigh Marshes are both used recreationally by walkers, particularly dog owners, and cyclists. The western half of Hadleigh Marsh embankment is owned by Essex County Council and managed in conjunction with the Environment Agency; it is part of Hadleigh Country Park and is occasionally used for livestock grazing (cows). The eastern half of the embankment is owned by the Salvation Army and managed in conjunction with the Environment Agency. Leigh Marshes has car parks, children's play areas, a golf driving range and a household waste recycling centre. With the exception of the golf driving range, Leigh Marshes is owned and managed by Southend Borough Council.

### 3.2.3 Designated areas within and near to the landfills

Both Hadleigh Marsh and Leigh Marshes landfills fall within a Ramsar, SPA, SSSI and a Marine Protected Area, and Leigh Marshes is also within a National Nature Reserve (Figure 3.4 to Figure 3.6). Both sites are within the bathing water catchments of eight public beaches on the Thames Estuary, and there are bivalve mollusc production areas located immediately downstream of the landfills (Figure 3.7).

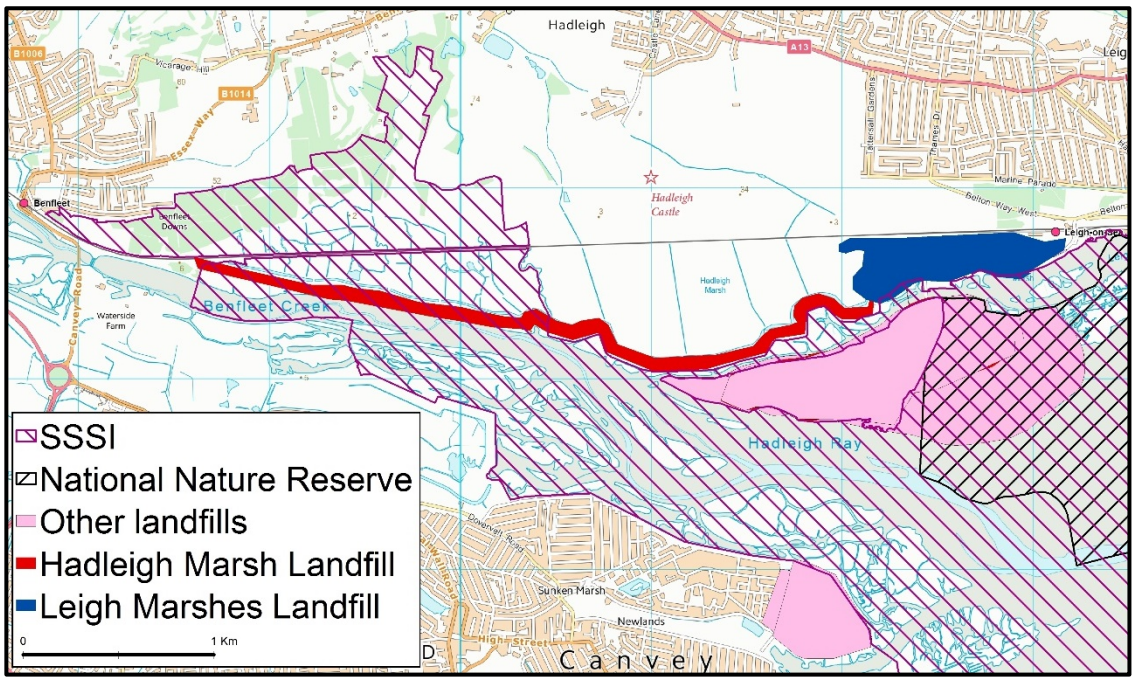



**Figure 3.4: Hadleigh Marsh and Leigh Marshes landfills and the adjacent estuary are partially designated as a SSSI, and Leigh Marshes and the estuary are also partially designated as National Nature Reserve (© Natural England copyright. Contains Ordnance Survey data © Crown copyright and database right 2014. © Crown copyright. All rights reserved. Natural England [2015]). (North up the page)**

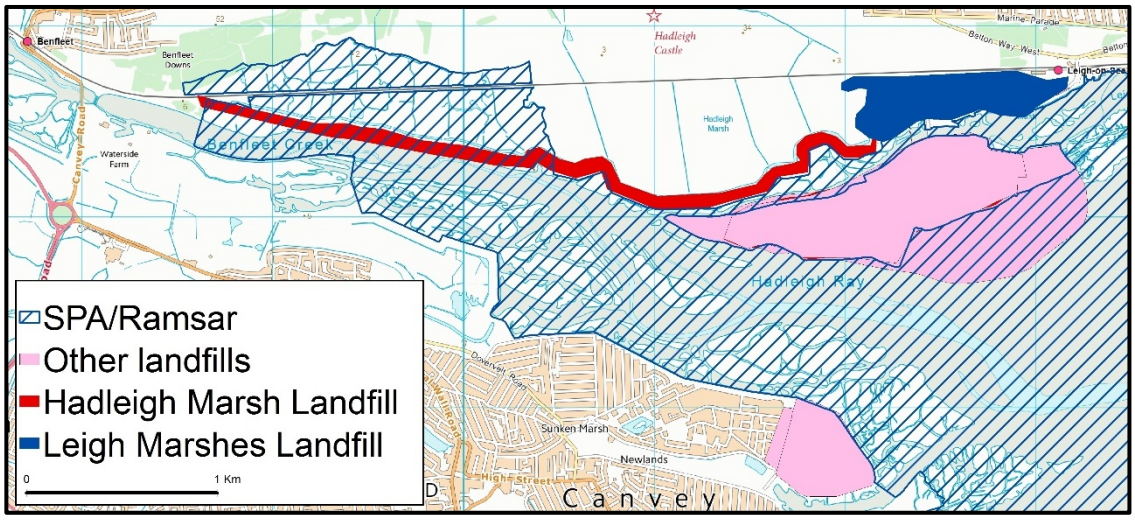



**Figure 3.5: Hadleigh Marsh and Leigh Marshes landfills and the adjacent estuary are partially designated as a Ramsar and SPA (© Natural England copyright. Contains Ordnance Survey data © Crown copyright and database right 2014. © Crown copyright. All rights reserved. Natural England [2015]). (North up the page)**

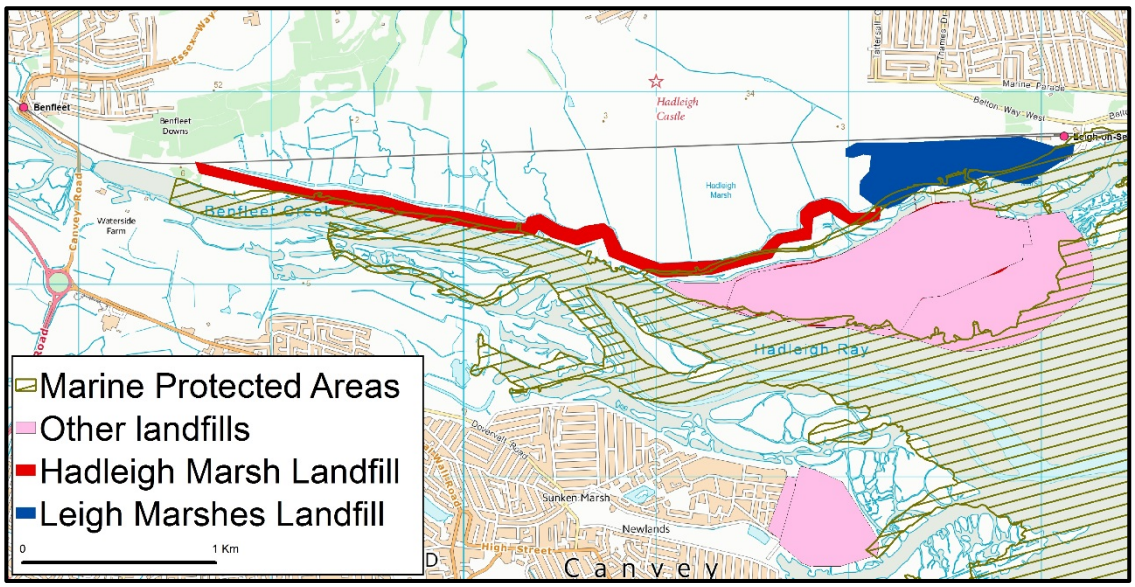



**Figure 3.6: Hadleigh Marsh and Leigh Marshes landfills and the adjacent estuary are partly designated as a Marine Protected Area (© Crown copyright. All rights reserved. Joint Nature Conservation Committee Support Co, 100017955 [2015]. Contains Ordnance Survey data © Crown copyright and database right 2014). (North up the page)**

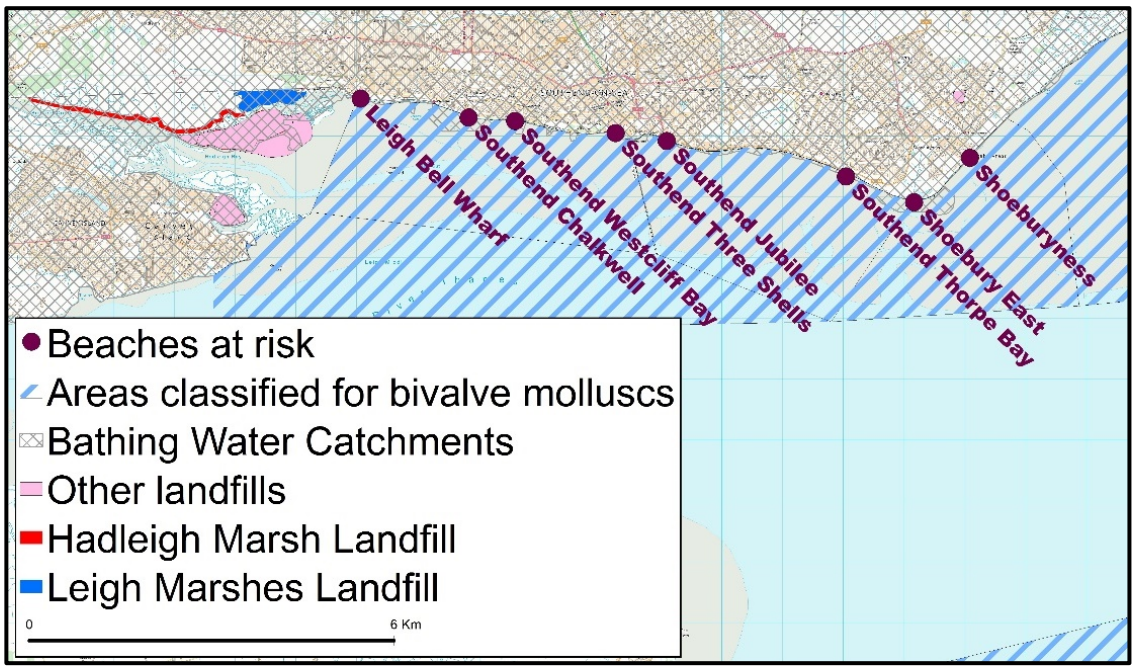



**Figure 3.7: Eight beaches at Shoeburyness, Leigh-on-Sea and Southend-on-Sea have Bathing Water Catchments that include Hadleigh Marsh and Leigh Marshes landfills, and bivalve mollusc production areas are also present downstream of the sites (Environment Agency, 2013a, contains Ordnance Survey data © Crown copyright and database right 2014, and bivalve production area data © Cefas, 2015). (North up the page)**

### 3.2.4 Previous monitoring of the sites

Essex County Council monitors water quality in the borrow ditch behind Hadleigh Marsh landfill quarterly. The parameters measured are ammoniacal nitrogen, chemical oxygen demand, chloride, dissolved oxygen, electrical conductivity, nitrate, nitrite and pH. The monitoring reports conclude that the ditch water is brackish and contains concentrations of ammoniacal nitrogen, nitrates and nitrites that can be attributed to the adjacent farmland (Caulmert Limited, 2012).

In 2012, the Environment Agency commissioned an extensive site investigation of Hadleigh Marsh and Leigh Marshes landfills (Halcrow Group Ltd, 2012). The investigation measured concentrations of As, Cd, Cr, Cu, Hg, Ni, Pb, Se and Zn sulphates, sulphides, cyanide, monohydric phenols, ammoniacal nitrogen, organic carbon, Polycyclic Aromatic Hydrocarbons (PAHs), aromatic hydrocarbons, aliphatic hydrocarbons, VOCs and SVOCs in matrix material samples taken from cores and trial pits. This research has identified mistakes in the calculations in spreadsheets used by Halcrow to obtain the statistical data published in its final report; so, this research has recalculated the statistical data using Halcrow's raw data (raw data supplied by R. Lancefield, pers. comm., email, 28/11/2013). In Leigh Marshes waste samples, only metals and PAHs had median concentrations above the limit of detection and have sediment quality guidelines available for comparison. In Hadleigh Marsh waste samples, only metals, PAHs, toluene, ethylbenzene, o-Xylene, styrene, 1,4-Dichlorobenzene and diethyl phthalate had median concentrations above the limit of detection and have sediment quality guidelines available for comparison. Metals and PAH concentrations are compared to Canadian Interim Sediment Quality Guidelines (ISQGs) and Probable Effect Levels (PELs) in Table 3.2 to Table 3.5 (Canadian Council of Ministers of the Environment, 2002), and the other contaminants, for which there are no Canadian guidelines, are compared to Dutch sediment quality guidelines in Table 3.6 (Buchman, 2008). In both Leigh Marshes and Hadleigh Marsh landfills metals and PAHs were found to be present at levels that exceed the PELs, above which adverse biological effects are frequently observed. The other contaminants exceeded Dutch sediment quality target values, i.e. the levels below which environmental risk is considered to be negligible, but did not exceed the intervention values, which are the

levels above which contaminants are considered to pose unacceptable environmental risk.

**Table 3.2: Metal concentrations determined by Halcrow in Leigh Marshes landfill compared to Canadian sediment quality guidelines (light grey = exceeds an ISQG, dark grey = exceeds a PEL, NB values that exceed PELs also exceed ISQGs)**

| | Canadian Fresh | | Canadian Marine | | Halcrow Leigh Marshes (n=30) | | | |
|---|---|---|---|---|---|---|---|---|
| | ISQG (mg $kg^{-1}$) | PEL (mg $kg^{-1}$) | ISQG (mg $kg^{-1}$) | PEL (mg $kg^{-1}$) | Min (mg $kg^{-1}$) | Max (mg $kg^{-1}$) | Mean (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
| As | 5.9 | 17 | 7.24 | 41.6 | 10 | 130 | 31 | 26 |
| Cd | 0.6 | 3.5 | 0.7 | 4.2 | <0.5 | 19 | 1.7 | 1.0 |
| Cr | 37.3 | 90 | 52.3 | 160 | 20 | 75 | 37 | 35 |
| Cu | 35.7 | 197 | 18.7 | 108 | 22 | 1191 | 248 | 145 |
| Hg | 0.17 | 0.486 | 0.13 | 0.7 | <0.5 | 3.1 | 0.8 | 0.6 |
| Ni | n/a | n/a | n/a | n/a | 16 | 177 | 46 | 39 |
| Pb | 35 | 91.3 | 30.2 | 112 | 34 | 3564 | 599 | 555 |
| Se | n/a | n/a | n/a | n/a | <0.5 | 2.5 | 1.2 | 1.1 |
| Zn | 123 | 315 | 124 | 271 | 72 | 1578 | 566 | 526 |

**Table 3.3: Metal concentrations determined by Halcrow in Hadleigh Marsh landfill compared to Canadian sediment quality guidelines (light grey = exceeds an ISQG, dark grey = exceeds a PEL, NB values that exceed PELs also exceed ISQGs)**

| | Canadian Fresh | | Canadian Marine | | Halcrow Hadleigh Marsh (n=25) | | | |
|---|---|---|---|---|---|---|---|---|
| | ISQG (mg $kg^{-1}$) | PEL (mg $kg^{-1}$) | ISQG (mg $kg^{-1}$) | PEL (mg $kg^{-1}$) | Min (mg $kg^{-1}$) | Max (mg $kg^{-1}$) | Mean (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
| As | 5.9 | 17 | 7.24 | 41.6 | <5 | 22 | 12 | 11 |
| Cd | 0.6 | 3.5 | 0.7 | 4.2 | <0.5 | 1.5 | | <0.5 |
| Cr | 37.3 | 90 | 52.3 | 160 | 8.5 | 86 | 29 | 27 |
| Cu | 35.7 | 197 | 18.7 | 108 | 13 | 232 | 42 | 30 |
| Hg | 0.17 | 0.486 | 0.13 | 0.7 | <0.5 | 19 | | <0.5 |
| Ni | n/a | n/a | n/a | n/a | 9.8 | 31 | 21.1 | 20 |
| Pb | 35 | 91.3 | 30.2 | 112 | 26.6 | 5340 | 331 | 106 |
| Se | n/a | n/a | n/a | n/a | <0.5 | 1.7 | | <0.5 |
| Zn | 123 | 315 | 124 | 271 | 53 | 41257 | 1801 | 125 |

**Table 3.4: PAHs determined by Halcrow in Leigh Marshes landfill compared to Canadian sediment quality guidelines (n=30, except 2-Methylnaphthalene: n=19) (light grey = exceeds an ISQG, dark grey = exceeds a PEL, NB values that exceed PELs also exceed ISQGs)**

| | Fresh ISQG (µg kg$^{-1}$) | Fresh PEL (µg kg$^{-1}$) | Marine ISQG (µg kg$^{-1}$) | Marine PEL (µg kg$^{-1}$) | Min (µg kg$^{-1}$) | Max (µg kg$^{-1}$) | Mean (µg kg$^{-1}$) | Median (µg kg$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| acenaphthene | 6.71 | 88.9 | 6.71 | 88.9 | <100 | 3897 | | <100 |
| acenaphthylene | 5.87 | 128 | 5.87 | 128 | <100 | 2454 | 342 | 113 |
| anthracene | 46.9 | 245 | 46.9 | 245 | <100 | 12114 | 1046 | 240 |
| benz(a)anthracene | 31.7 | 385 | 74.8 | 693 | <100 | 29157 | 3269 | 1281 |
| benzo(a)pyrene | 31.9 | 782 | 88.8 | 763 | <100 | 28240 | 3308 | 1250 |
| benzo(b)fluoranthene | | | | | <100 | 24723 | 3163 | 1279 |
| benzo(ghi)perylene | | | | | <100 | 18032 | 2784 | 1283 |
| benzo(k)fluoranthene | | | | | <100 | 21935 | 2888 | 1065 |
| chrysene | 57.1 | 862 | 108 | 846 | <100 | 30234 | 3492 | 924 |
| dibenz(ah)anthracene | 6.22 | 135 | 6.22 | 135 | <100 | 4688 | 687 | 304 |
| fluoranthene | 111 | 2355 | 113 | 1494 | <100 | 65074 | 6151 | 1593 |
| fluorene | 21.2 | 144 | 21.2 | 144 | <100 | 4294 | 381 | 168 |
| indeno(123-cd)pyrene | | | | | <100 | 19978 | 2952 | 1340 |
| 2-methylnaphthalene | 20.2 | 201 | 20.2 | 201 | <10 | 57840 | 3085 | 24 |
| naphthalene | 34.6 | 391 | 34.6 | 391 | <100 | 1207 | | <100 |
| phenanthrene | 41.9 | 515 | 86.7 | 544 | <100 | 43077 | 3112 | 459 |
| pyrene | 53 | 875 | 153 | 1398 | <100 | 52490 | 5135 | 1752 |

**Table 3.5: PAHs determined by Halcrow in Hadleigh Marsh landfill compared to Canadian sediment quality guidelines (n=25, except 2-Methylnaphthalene: n=11) (light grey = exceeds an ISQG, dark grey = exceeds a PEL, NB values that exceed PELs also exceed ISQGs)**

| | Fresh ISQG (µg kg$^{-1}$) | Fresh PEL (µg kg$^{-1}$) | Marine ISQG (µg kg$^{-1}$) | Marine PEL (µg kg$^{-1}$) | Min (µg kg$^{-1}$) | Max (µg kg$^{-1}$) | Mean (µg kg$^{-1}$) | Median (µg kg$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| acenaphthene | 6.71 | 88.9 | 6.71 | 88.9 | <100 | 4311 | | <100 |
| acenaphthylene | 5.87 | 128 | 5.87 | 128 | <100 | 967 | | <100 |
| anthracene | 46.9 | 245 | 46.9 | 245 | <100 | 3504 | | <100 |
| benz(a)anthracene | 31.7 | 385 | 74.8 | 693 | <100 | 6655 | 1213 | 181 |
| benzo(a)pyrene | 31.9 | 782 | 88.8 | 763 | <100 | 4962 | 1019 | 153 |
| benzo(b)fluoranthene | | | | | <100 | 4900 | 874 | 195 |
| benzo(ghi)perylene | | | | | <100 | 3439 | 745 | 130 |
| benzo(k)fluoranthene | | | | | <100 | 8296 | 1138 | 130 |
| chrysene | 57.1 | 862 | 108 | 846 | <100 | 6410 | 1268 | 286 |
| dibenz(ah)anthracene | 6.22 | 135 | 6.22 | 135 | <100 | 988 | | <100 |
| fluoranthene | 111 | 2355 | 113 | 1494 | <100 | 15213 | 2869 | 358 |
| fluorene | 21.2 | 144 | 21.2 | 144 | <100 | 3999 | 520 | 166 |
| indeno(123-cd)pyrene | | | | | <100 | 4549 | 780 | 186 |
| 2-methylnaphthalene | 20.2 | 201 | 20.2 | 201 | <10 | 3879 | 515 | 199 |
| naphthalene | 34.6 | 391 | 34.6 | 391 | <100 | 4572 | | <100 |
| phenanthrene | 41.9 | 515 | 86.7 | 544 | <100 | 11052 | 2451 | 539 |
| pyrene | 53 | 875 | 153 | 1398 | <100 | 11988 | 2238 | 346 |

**Table 3.6: Other organic contaminants determined by Halcrow in Hadleigh Marsh landfill with median concentrations greater than the limit of detection and that have sediment quality guidelines available for comparison (n=11) (light grey = exceeds target, none exceed intervention)**

| | Dutch target (µg kg$^{-1}$) | Dutch intervention (µg kg$^{-1}$) | Min (µg kg$^{-1}$) | Max (µg kg$^{-1}$) | Mean (µg kg$^{-1}$) | Median (µg kg$^{-1}$) |
|---|---|---|---|---|---|---|
| toluene | 10 | 47000 | <10 | 131 | 48 | 53 |
| ethylbenzene | 30 | 50000 | <10 | 267 | 98 | 62 |
| o-xylene | 89 | 9300 | <10 | 198 | 69 | 48 |
| styrene | 200 | 86000 | <10 | 61 | 23 | 15 |
| 1,4-dichlorobenzene | 30 | 18000 | <10 | 3940 | 577 | 40 |
| diethyl phthalate | 530 | 53000 | <10 | 1141 | 266 | 66 |

## 3.3 Geophysical site investigation methods

Geophysical ground investigation techniques were used to investigate the size of the waste cells within Hadleigh Marsh landfill and hence the likely maximum extent of any breach of the embankment. Electromagnetic techniques, Ground Penetrating Radar (GPR – Sensors & Software Inc. pulseEKKO Pro Model 1100) and ground conductivity meter (Geonics EM31), were chosen as they are suitable for investigating large sites quickly and at low cost (Milsom and Eriksen, 2011; Cassidy et al., 2014) and EM31 has previously been used to determine the boundaries of landfill sites (Green et al., 1999).

A GPR trial was carried out on 18th March 2013. The GPR's signal was unable to penetrate the clay capping of the landfill and produced no usable data. This was a known risk as clay and saltwater, which the clay capping is likely to contain, can rapidly attenuate radar signals (Meju, 2000; Milsom and Eriksen, 2011), in addition landfill waste itself highly attenuates radio waves (Reynolds, 2011).

The EM31 survey was carried out on 17th December 2014. The complete length of the embankment (from west to east) and seven transverse sections were surveyed to determine longitudinal and transverse features in the landfill structure. The scan along the length of the embankment was approximately centred on the top of the embankment. The transverse survey locations were randomly chosen on site at locations where it was deemed safe to walk down the sides of the embankment, i.e. no slip/trip hazards. Approximately 600 metres at the start of the survey and one transverse section were not captured due to a data logger error which was not identified until the positioning data was transferred into ArcMap.

The EM31 data were downloaded and processed by Dr. Martin Smith from the University of Brighton who also advised upon data interpretation to identify which signal variations were significant and the likely causes for those signal variations.

## 3.4 Sample collection methods

Attempts to obtain samples from Hadleigh Marsh through manual augering in November 2012 and windowless sampling on the 25th November 2013 were unsuccessful due to the dense clay capping covering the site and the densely packed waste materials impeding ground penetration. It was therefore necessary to mechanically excavate trial pits to obtain samples. Trial pit locations were selected to avoid locations at which previous sampling had been undertaken in 2012 by Halcrow (Halcrow Group Ltd, 2012), to minimise the environmental impact on the designated sites and to avoid services. Six locations (see Table 3.7 and Figure 3.1) were agreed with the stakeholders (see Table 3.8), two of which were contingencies (trial pits 1 and 2) in case the preferred locations were deemed unsuitable by the supervising ecologist during the works. The presence of an ecologist was mandatory to obtain Natural England assent for the works. Assent was required due to the sites' designations and the invasive nature of the works in an area known to contain slowworms and which has the potential for other reptiles and subterranean Shrill Carder Bee (*Bombus sylvarum*) nests to be present.

**Table 3.7: Trial pit location coordinates**

| Trial pit | Approximate OSGB36 National Grid Reference | Easting | Northing |
|---|---|---|---|
| 1 | TQ 79892 85381 | 579892 | 185381 |
| 2 | TQ 80345 85307 | 580345 | 185307 |
| 3 | TQ 80964 85087 | 580964 | 185087 |
| 4 | TQ 81688 85239 | 581688 | 185239 |
| 5 | TQ 82534 85649 | 582534 | 185649 |
| 6 | TQ 82640 85683 | 582640 | 185683 |

**Table 3.8: Hadleigh Marsh and Leigh Marshes site stakeholders**

| Stakeholder | Interest in sites |
|---|---|
| Environment Agency | Flood defence/Flood plain |
| Essex County Council | Landowners and landfill managers (western half of Hadleigh Marsh and the Country Park and arable land it protects) |
| Natural England | Ramsar/SSSI/SPA and suitable habitat for protected species |
| Salvation Army | Landowners (eastern half of Hadleigh Marsh and land it protects) |
| Southend Borough Council | Landowners (Leigh Marshes) |

Samples were collected in March 2014 from four excavator dug trial pits (Plate 3.3), two at each study site. The works were timed to avoid reptile hibernation periods, which subject to weather conditions are typically from early November to late February (CIRIA, 2011), and to meet the terms of the Natural England assent, which required works to be completed by the 31st of March 2014. The trial pits were excavated at locations 3, 4, 5 and 6 listed in Table 3.7 and shown in Figure 3.1. Trial pit 6 was dug on the 26th March, the other trial pits on the 27th March. There was light rain during the collection of samples from trial pit 6, there was no rain during the collection of the other samples. Samples were obtained from a maximum depth of 2.1 metres in Leigh Marshes (capping layer 0.50-0.90 metres depth) and 1.5 metres in Hadleigh Marsh (capping layer 0.85-0.90 metres depth). The sides of the trial pits were unstable below the capping layer, which prohibited sampling from greater depths.


**Plate 3.3: Excavator sampling, Hadleigh Marsh embankment, March 2014**

At each trial pit approximately 360 litres of waste was collected in 8 x 45 litre plastic containers. These were arranged in a four by two grid within a skip bag (to contain any overspill, Figure 3.8) and waste samples were randomly dropped into them directly from the excavator bucket. The excavator bucket was jet washed between each sampling location to minimise the risk of cross-contamination. Organic contaminants are known to sorb to plastics, therefore, four containers for each trial pit were foil lined (Karapanagioti et al., 2009). The containers were labelled with the trial pit number and a letter; letters A to D were used for the unlined sample containers and E to H for the foil

lined sample containers. The containers were sealed to maintain field moisture levels and enable safe transportation; in addition to the chemical hazards associated with landfill waste, the same types and counts of pathogens can be found in household waste as hospital waste (Jager et al., 1989; cited in Lisk, 1991), including some that can survive for at least 9 years in landfills (Donnelly et al., 1982; Pahren and Clark, 1987), and leachate can contain anti-biotic resistant bacteria (Wang et al., 2015a). The containers were transported to QMUL's laboratory where samples collected for inorganic contaminant analysis were refrigerated and samples collected for organic contaminant analysis were frozen until analyses were undertaken.

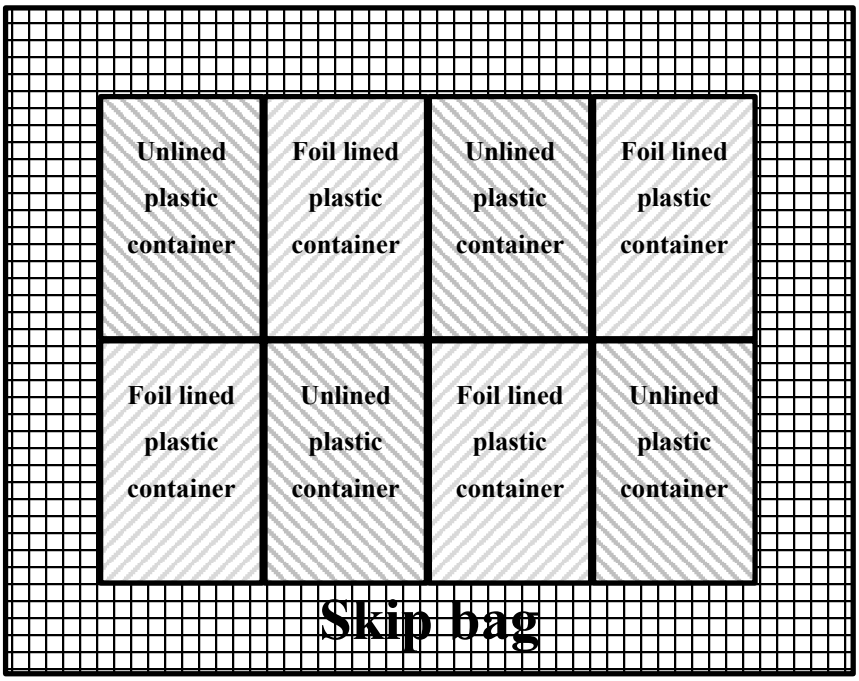



**Figure 3.8: Schematic showing the arrangement of sampling containers within the skip bag at the time of sample collection**

## *3.5 Contaminant analysis methods*

### 3.5.1 Sample screening

Due to the scarcity of records and hence limited knowledge regarding the contaminants present, one organic and one inorganic subsample of mixed materials for each trial pit were sent to the EA National Laboratory to provide initial screening data and identify contaminants of concern. The results of the screening analysis are compared in Table 3.9 and Table 3.10 to Canadian Sediment Quality Guidelines for the protection of aquatic life (Canadian Council of Ministers of the Environment, 2002), which are commonly used in the absence of UK sediment quality guidelines. The majority of contaminants exceed the Interim Sediment Quality Guidelines (ISQGs), below which adverse biological effects are expected to rarely occur, and many of the contaminant levels exceed the Probable Effect Levels (PELs), above which adverse effects are frequently observed. These results confirmed that metal and PAH concentrations in the waste samples warranted further investigation.

**Table 3.9: Contaminant levels in screening samples vs. Canadian freshwater sediment guidelines, where available (light grey > ISQG, dark grey > PEL, NB values that exceed PELs also exceed ISQGs)**

| | | Canadian Freshwater Sediment Guidelines | | Hadleigh Marsh | | Leigh Marshes | |
|---|---|---|---|---|---|---|---|
| | Units | ISQG | PEL | Trial Pit 3 | Trial Pit 4 | Trial Pit 5 | Trial Pit 6 |
| As | mg $kg^{-1}$ | 5.9 | 17 | 19 | 12 | 135 | 72.8 |
| Ba | mg $kg^{-1}$ | | | 290 | 493 | 119 | 77 |
| B | mg $kg^{-1}$ | | | 44.2 | 40.3 | 127 | 83.8 |
| Cd | mg $kg^{-1}$ | 0.6 | 3.5 | 1.53 | 4.62 | 1.51 | 7.84 |
| Cr | mg $kg^{-1}$ | 37.3 | 90 | 165 | 70 | 72.5 | 82.5 |
| Cu | mg $kg^{-1}$ | 35.7 | 197 | 117 | 97.7 | 753 | 340 |
| Pb | mg $kg^{-1}$ | 35 | 91.3 | 643 | 162 | 1390 | 1220 |
| Hg | mg $kg^{-1}$ | 0.17 | 0.486 | <0.2 | <0.2 | 1.09 | 2.74 |
| Ni | mg $kg^{-1}$ | | | 54.6 | 46.3 | 141 | 121 |
| Se | mg $kg^{-1}$ | | | <1 | <1 | 1.02 | <1 |
| Zn | mg $kg^{-1}$ | 123 | 315 | 3080 | 435 | 1840 | 3010 |
| acenaphthene | µg $kg^{-1}$ | 6.71 | 88.9 | 97.8 | 10.6 | 69.5 | 938 |
| acenaphthylene | µg $kg^{-1}$ | 5.87 | 128 | 135 | 12.8 | 63.3 | 88.3 |
| anthracene | µg $kg^{-1}$ | 46.9 | 245 | 343 | 31.4 | 384 | 3210 |
| benzo(a)anthracene | µg $kg^{-1}$ | 31.7 | 385 | 2130 | 200 | 1910 | 13000 |
| benzo(a)pyrene | µg $kg^{-1}$ | 31.9 | 782 | 2510 | 223 | 1940 | 10100 |
| benzo(b)fluoranthene | µg $kg^{-1}$ | | | 2840 | 337 | 2580 | 15700 |
| benzo(ghi)perylene | µg $kg^{-1}$ | | | 1660 | 223 | 1250 | 5790 |
| benzo(k)fluoranthene | µg $kg^{-1}$ | | | 966 | 114 | 773 | 5040 |
| chrysene | µg $kg^{-1}$ | 57.1 | 862 | 2460 | 300 | 2320 | 14600 |
| coronene | µg $kg^{-1}$ | | | 478 | 74.4 | 330 | 1140 |
| dibenzo(ah)anthracene | µg $kg^{-1}$ | 6.22 | 135 | 494 | 61.3 | 467 | 2280 |
| fluoranthene | µg $kg^{-1}$ | 111 | 2355 | 4430 | 424 | 4060 | 20400 |
| fluorene | µg $kg^{-1}$ | 21.2 | 144 | 141 | 21.5 | 96.2 | 1500 |
| indeno(1,2,3-cd)pyrene | µg $kg^{-1}$ | | | 2110 | 269 | 1800 | 8480 |
| naphthalene | µg $kg^{-1}$ | 34.6 | 391 | 163 | 48 | 546 | 7650 |
| phenanthrene | µg $kg^{-1}$ | 41.9 | 515 | 1380 | 199 | 2020 | 11500 |
| pyrene | µg $kg^{-1}$ | 53 | 875 | 4040 | 442 | 3590 | 14900 |

**Table 3.10: Contaminant levels in screening samples vs. Canadian marine sediment guidelines, where available (light grey > ISQG, dark grey > PEL, NB values that exceed PELs also exceed ISQGs)**

| | | Canadian Marine Sediment Guidelines | | Hadleigh Marsh | | Leigh Marshes | |
|---|---|---|---|---|---|---|---|
| | Units | ISQG | PEL | Trial Pit 3 | Trial Pit 4 | Trial Pit 5 | Trial Pit 6 |
| As | mg $kg^{-1}$ | 7.24 | 41.6 | 19 | 12 | 135 | 72.8 |
| Ba | mg $kg^{-1}$ | | | 290 | 493 | 119 | 77 |
| B | mg $kg^{-1}$ | | | 44.2 | 40.3 | 127 | 83.8 |
| Cd | mg $kg^{-1}$ | 0.7 | 4.2 | 1.53 | 4.62 | 1.51 | 7.84 |
| Cr | mg $kg^{-1}$ | 52.3 | 160 | 165 | 70 | 72.5 | 82.5 |
| Cu | mg $kg^{-1}$ | 18.7 | 108 | 117 | 97.7 | 753 | 340 |
| Pb | mg $kg^{-1}$ | 30.2 | 112 | 643 | 162 | 1390 | 1220 |
| Hg | mg $kg^{-1}$ | 0.13 | 0.7 | <0.2 | <0.2 | 1.09 | 2.74 |
| Ni | mg $kg^{-1}$ | | | 54.6 | 46.3 | 141 | 121 |
| Se | mg $kg^{-1}$ | | | <1 | <1 | 1.02 | <1 |
| Zn | mg $kg^{-1}$ | 124 | 271 | 3080 | 435 | 1840 | 3010 |
| acenaphthene | µg $kg^{-1}$ | 6.71 | 88.9 | 97.8 | 10.6 | 69.5 | 938 |
| acenaphthylene | µg $kg^{-1}$ | 5.87 | 128 | 135 | 12.8 | 63.3 | 88.3 |
| anthracene | µg $kg^{-1}$ | 46.9 | 245 | 343 | 31.4 | 384 | 3210 |
| benzo(a)anthracene | µg $kg^{-1}$ | 74.8 | 693 | 2130 | 200 | 1910 | 13000 |
| benzo(a)pyrene | µg $kg^{-1}$ | 88.8 | 763 | 2510 | 223 | 1940 | 10100 |
| benzo(b)fluoranthene | µg $kg^{-1}$ | | | 2840 | 337 | 2580 | 15700 |
| benzo(ghi)perylene | µg $kg^{-1}$ | | | 1660 | 223 | 1250 | 5790 |
| benzo(k)fluoranthene | µg $kg^{-1}$ | | | 966 | 114 | 773 | 5040 |
| chrysene | µg $kg^{-1}$ | 108 | 846 | 2460 | 300 | 2320 | 14600 |
| coronene | µg $kg^{-1}$ | | | 478 | 74.4 | 330 | 1140 |
| dibenzo(ah)anthracene | µg $kg^{-1}$ | 6.22 | 135 | 494 | 61.3 | 467 | 2280 |
| fluoranthene | µg $kg^{-1}$ | 113 | 1494 | 4430 | 424 | 4060 | 20400 |
| fluorene | µg $kg^{-1}$ | 21.2 | 144 | 141 | 21.5 | 96.2 | 1500 |
| indeno(1,2,3-cd)pyrene | µg $kg^{-1}$ | | | 2110 | 269 | 1800 | 8480 |
| naphthalene | µg $kg^{-1}$ | 34.6 | 391 | 163 | 48 | 546 | 7650 |
| phenanthrene | µg $kg^{-1}$ | 86.7 | 544 | 1380 | 199 | 2020 | 11500 |
| pyrene | µg $kg^{-1}$ | 153 | 1398 | 4040 | 442 | 3590 | 14900 |

### 3.5.2 Sorting materials for extraction

The matrix material, textiles and paper were selected for analysis due to their large specific surface area, which correlates with inorganic contaminant load (Parizanganeh, 2007), and the fact that they typically make up a significant proportion of household waste (Figure 3.2). Wood was additionally selected as, although it does not typically make up a significant proportion of household waste, it was prevalent in the waste samples excavated from Hadleigh Marsh landfill, possibly due to the site also containing commercial waste.

All handling of waste and material sorting was carried out in a fume cupboard. The large volume of waste collected, circa 360 litres per trial pit, and handling restrictions made it impracticable to sort all materials to determine the total mass or volume of each material type in the samples. For Leigh Marshes and Hadleigh Marsh landfills subsamples of matrix material were obtained from each inorganic sample container by sieving approximately 100 g of the <2 mm fraction of waste into plastic bags. Material over 2 mm was returned to the sample container. Between containers the sieve was washed, rinsed with deionised water and dried. The <2 mm fraction of the waste was selected for analysis as inorganic contaminants sorb to the smallest particles in the greatest concentrations (Parizanganeh, 2007) and it allowed comparison to Halcrow's (2012) previous study of the matrix material (Environmental Laboratory Ltd, pers. comm., email, 6/11/2014).

In addition, for Hadleigh Marsh landfill, subsamples of textiles, paper and wood were obtained from each inorganic sample container that contained sufficient material (Table 3.11) by sieving approximately 100 g of the <10 mm fraction of waste into plastic bags. These materials were cut from larger samples where necessary and care was taken to select samples with minimal quantities of attached matrix material, e.g. the outer pages of newspapers were not used. The <10 mm fraction was used for textiles, paper and wood as it was impracticable to sieve these materials to <2 mm (Riber et al., 2007). Leigh Marshes did not contain these materials in sufficient quantities to enable analysis.

**Table 3.11: Materials selected for extractions from the Hadleigh Marsh landfill sample containers**

| Trial pit number | Container code | Matrix | Wood | Paper | Textiles |
|---|---|---|---|---|---|
| 3 | 3A | ✓ | ✓ | ✓ | |
| 3 | 3B | ✓ | ✓ | | ✓ |
| 3 | 3C | ✓ | ✓ | ✓ | |
| 3 | 3D | ✓ | ✓ | ✓ | |
| 4 | 4A | ✓ | | ✓ | |
| 4 | 4B | ✓ | | | ✓ |
| 4 | 4C | ✓ | | ✓ | |
| 4 | 4D | ✓ | ✓ | ✓ | ✓ |

### 3.5.3 Moisture content

For each of the inorganic sample containers, a minimum of three subsamples of approximately 10 g of matrix material, sieved to <2 mm, were weighed into pre-weighed crucibles. The crucibles were acid washed (10% $HNO_3$), rinsed three times in deionised water and dried before use to enable the samples to be used later for analysis of major and trace metals (see section 3.5.4). Health and safety restrictions required the waste to be in sealed containers when not in a fume cupboard, it was therefore not possible to use the standard method of oven drying the samples; instead the samples were air dried overnight and then on a hotplate for 6 hours at 105 °C in a fume cupboard. The drying time used was determined by trial runs showing no further decrease in mass of samples between 6 hours and 7 hours of drying when measured to an accuracy of 10 mg. The dried samples were then reweighed and the initial moisture content calculated on a dry mass basis (Equation 3.1).

**Equation 3.1: Calculation of moisture content in waste materials on a dry mass basis (ISO, 1993)**

$$Moisture\ content\ (\%) = \left(\frac{wet\ mass\ of\ material - dry\ mass\ of\ material}{dry\ mass\ of\ material}\right) x\ 100\%$$

In addition, for the Hadleigh Marsh samples, three subsamples of textiles, paper and wood sieved to <10 mm were dried from each container that had sufficient quantities of the material of interest using the same method, totalling 15 wood, 18 paper and 9 textile samples dried.

### 3.5.4 Inorganic contaminant analysis

**Metal extractions**

Concentrations of a suite of major and trace metals were determined in the matrix material, wood, textiles and paper samples using hotplate aqua regia extractions of dried subsamples (after Chen and Ma, 2001). The extracts were then analysed using Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) and Inductively Coupled Plasma Mass Spectrometry (ICP-MS). This extraction method was chosen as it determines a pseudo-total contaminant concentration that enables comparison to commonly used environmental quality guidelines for sediment (Canadian Council of Ministers of the Environment, 2001).

Equipment used for the extractions was acid washed (10% $HNO_3$), rinsed three times in deionised water and dried before use. 0.5 (± 0.025) g subsamples of dried waste which had first been sieved to <2 mm (matrix material) or cut up and sieved to <10 mm (textiles/paper/wood), were heated in 12 ml aqua regia (3:1 HCl:$HNO_3$) for 5 hours. The resulting solutions were then filtered (using Whatman 542 filter papers) into volumetric flasks and diluted with deionised water to a total volume of 50 ml.

Three samples for extraction were taken from each of the dried matrix material, wood, paper and textile subsamples. This totalled 72 matrix material extractions per site, two were spilt leaving 71 per site for analysis, and 45 wood extractions, 54 paper extractions and 27 textile extractions for Hadleigh Marsh landfill. Following ICP-OES analysis there was insufficient sample remaining of one paper extraction to allow ICP-MS analysis, therefore data for 53 paper extractions are reported.

In order to test the method accuracy, certified reference materials (BCR143R, LGC6137 and LGC6187) were extracted alongside the subsamples; mean recoveries ranged from 85-122% with an average relative standard deviation (%RSD) of 11%. Method precision was tested using replicate extractions of the certified reference materials (CRMs) and achieved an average %RSD of 11.8%. 10% blanks were used to check for background contamination, and concentrations of metals reported for the waste samples were adjusted accordingly.

**Metal analysis – ICP OES**

Metal extractions were first analysed using ICP-OES (Agilent Technologies Vista Pro). A multi-element standard, containing Ag, Al, B, Ba, Ca, Cd, Co, Cr, Cu, Fe, K, Li, Mg, Mn, Na, Ni, Pb, Sr and Zn, was used to make matrix matched standards of 0, 0.2, 0.6, 1, 5, 10 and 50 mg $L^{-1}$ to calibrate the ICP-OES, and the 5 mg $L^{-1}$ standard was used as the laboratory control sample to monitor instrument drift during analysis. Matrix matching was achieved by creating standards using 24% aqua regia. Where metal concentrations in the samples were found to exceed the calibrated range of the ICP-OES, samples were diluted and then re-analysed. The samples were diluted using a 24% aqua regia:76% deionised water solution to maintain the proportion of acid for matrix matching.

ICP-OES instrument precision was tested using analysis of randomly selected samples in triplicate and achieved an average %RSD of 6.2%.

In some samples, Ag, Ba, Cd, Co, Li, Ni and Pb were below the limit of detection (LOD) for ICP-OES. A random selection were analysed by both AAS and ICP-MS to determine whether they would be suitable alternative analysis methods. The samples were also below LOD for AAS, but above LOD for ICP-MS, with the exception of Ag, which was below LOD for both. Silver is not considered a significant environmental pollutant, there are no sediment quality standards or guidelines for it; therefore, Ag was not considered further.

**Metal analysis – ICP-MS**

Concentrations of Cd in the matrix material extractions for trial pits 3, 4 and 5 were determined by ICP-MS, reported data for the other metals in the matrix material are from ICP-OES. Paper is likely to be present in Hadleigh Marsh landfill in at least the same quantities (by mass[i]) as the matrix material and, therefore, the paper samples were also analysed by ICP-MS as the concentrations of Ba, Cd, Co, Li, Ni and Pb were below ICP-OES LOD. All reported metal data for paper are from ICP-MS analysis. Wood and textile samples with contaminant concentrations below the ICP-OES limit of detection were not reanalysed using ICP-MS because the significantly higher quantities of matrix

[i] Figure 3.2 shows that the mass paper is approximately the same as that of matrix material (fines+misc+putrescible materials) in typical household waste (before drying), but when the dry mass is calculated (Equation 3.3) it is likely there is circa 50% more paper than matrix material; however, since the site was constructed some paper may have degraded and integrated into the matrix material

material and paper typically found within waste means these materials would not significantly contribute to the total load of those contaminants within the waste.

ICP-MS (Thermo Scientific X Series 2) analysis was carried out by Plymouth University using undiluted samples. The method and instrument precision were tested together through analysis of replicate extractions of the CRMs and achieved an average %RSD of 6.7%.

### 3.5.5 Organic contaminant analysis

Initial investigations into waste heterogeneity using inorganic contaminant data indicated that the waste within the sites is highly variable, requiring large numbers of samples to be analysed to provide representative data (see section 3.8). This would have been prohibitively expensive for the analysis of multiple organic contaminants. A review of existing data for the site (see section 3.2.4) found most organic contaminants were below detection and that PAHs pose the greatest ecological risk at both Leigh Marshes and Hadleigh Marsh landfills. Therefore, PAHs were chosen as the focus for the organic contaminant element of this research as they are persistent organic pollutants and gaining a detailed understanding of how they are distributed would provide the most beneficial data for understanding the potential for pollution should these sites erode.

Subsamples of the matrix material from both sites were sent for analysis at the EA National Laboratory to determine concentrations of PAHs. The matrix material was selected for analysis due to its potential to contain or sorb PAHs and its presence in both sites in significant quantities. Wood, paper and textiles are not suitable for commercially available PAH analysis methods and, therefore, were not analysed. Three matrix material subsamples of approximately 250 ml from each of the organic sample containers that contained sufficient matrix material were sent, totalling 21 samples for Hadleigh Marsh and 24 for Leigh Marshes. The samples were packaged in dark glass jars with PTFE lids then double wrapped in sealed plastic bags before being shipped in corrugated plastic boxes. One Hadleigh Marsh sample was deemed unsuitable for analysis by the EA National Laboratory as they were unable to sieve it to the required fraction, leaving 20 samples analysed for Hadleigh Marsh. The analytical protocol for

PAH analysis used by the EA National Laboratory is as follows (National Laboratory Service, nd):

1. Samples are crushed using a jaw crusher and sieved to <2 mm fraction.
2. Samples are air-dried at <30 °C until constant weight is achieved.
3. Air-dried samples are extracted in dichloromethane for 12 to 24 hours in a vial at room temperature. A portion of the solvent layer is removed and solvent exchanged into hexane cleaned up by solid phase extraction and analysed on a GC-MS system.

### 3.5.6 Calculation of the total load of a contaminant in a landfill site

In order to understand which materials are important to analyse when investigating landfill contamination it is necessary to understand the contribution of different materials to the total contaminant load of the landfill. Equation 3.2 shows the method used for calculating the total load of a contaminant in a specific material type based on typical proportions of materials in household waste when the landfill was constructed. As no data exist that enable determination of the proportions of household, industrial and commercial wastes in the sites or the typical materials in industrial or commercial wastes, it has been assumed that they are similar in constitution to household waste. This may overestimate the proportion of matrix material in the sites and underestimate the proportion of paper as commercial waste typically has a higher paper content and lower organic content that household waste (Burnley, 2007a), but is the best available data to evaluate the relative contributions of different materials to the total contaminant load. Wood could not be included in the calculations as no data exist relating to the proportion of wood in landfills. However, this is unlikely to be important as the matrix material and paper are expected to contribute the majority of the total contaminant load of landfill sites (Flyhammar, 1997) as they constitute the majority of materials in the site that are likely to have high contaminant loads due to their large specific surface area to volume ratios (Parizanganeh, 2007).

**Equation 3.2: Calculation of the total contaminant load of a specific material type**

$$Total\ contaminant\ load\ of\ a\ specific\ material\ type\ (kg) = m\ x\ C\ x\ 0.001$$

*Where:*
*$m$ = dry mass of the material type of interest (tonnes), see Equation 3.3*
*$C$ = median concentration of the contaminant in the material type (mg kg$^{-1}$)*

**Equation 3.3: Calculation of the dry mass of the material type of interest NB to obtain the dry mass of the matrix material the dry masses of fines, misc. and putrescible materials must be calculated individually and summed**

$$m = Vx\ D\ x\ \frac{P}{100}\ x\ (1 - \frac{L_c}{100})$$

*Where:*
*V = total volume of landfill waste at site of interest ($m^3$)*
*D = typical landfill density at construction, 0.54-0.72 tonnes $m^{-3}$ (Leonard Sr et al., 2000)*
*P = percentage of typical household waste that was the material of interest at the time the landfill was constructed, estimated from Figure 3.2.*
*$L_c$ = moisture content (%) in material at time of landfilling. Wood=20%, paper=8%, textiles=10%, fines and misc.=8%, putrescible materials=70% (Tchobanoglous et al., 1993; Riber et al., 2009)*

### 3.5.7 Statistical methods

Where more than 50% of the results for a contaminant were over the Limit of Detection (LOD), values below the LOD were replaced (censored) with a value of 0.5 x LOD (Verbovsek, 2011). Where 50% or more of the results for a contaminant were below the LOD (i.e. median < LOD) median values are not reported. When carrying out statistical comparisons, when median values were below the LOD for both datasets statistical data were not calculated. Nonparametric statistical methods were used throughout this thesis as the majority of the contaminant datasets were not normally distributed (determined using frequency distributions and Kolmogorov-Smirnov tests). In addition, nonparametric methods are more robust than parametric methods when datasets include censored values (Verbovsek, 2011). In the correlation analyses the strength of correlation reported was based on the ranges shown in Table 3.12. All statistical analyses were undertaken using IBM SPSS Statistics 22. Full datasets can be found in Appendix A.

**Table 3.12: Strength of correlation (after Devore and Peck, 2001)**

| Correlation coefficient | Strength of correlation |
|---|---|
| -1 ≥ coefficient ≤ -0.8 | Strong negative |
| -0.8 > coefficient ≤ -0.5 | Moderate negative |
| -0.5 > coefficient < 0 | Weak negative |
| 0 ≤ coefficient < 0.5 | Weak positive |
| 0.5 ≤ coefficient < 0.8 | Moderate positive |
| 0.8 ≤ coefficient ≤ 1 | Strong positive |

## 3.6 Geophysical survey results

The outputs from the EM31 survey carried out along the length of Hadleigh Marsh landfill embankment are shown in Figure 3.9. The In-phase signal, which can indicate the presence of metallic objects (Reynolds International, 2011), was consistently 20.48 ppt along the embankment with the exception of three locations, where it dropped to -20.47 ppt. In contrast, the conductivity continuously fluctuated along the length of the embankment, ranging from -204.7 to 151.1 mS $m^{-1}$, with a mean value of 108.5 mS $m^{-1}$ and a median value of 112.0 mS $m^{-1}$. The outputs from the transverse EM31 surveys and their locations are shown in Figure 3.10, the conductivity was highest at the lowest points of the embankment, and the In-phase signal did not vary across the sections or between locations.

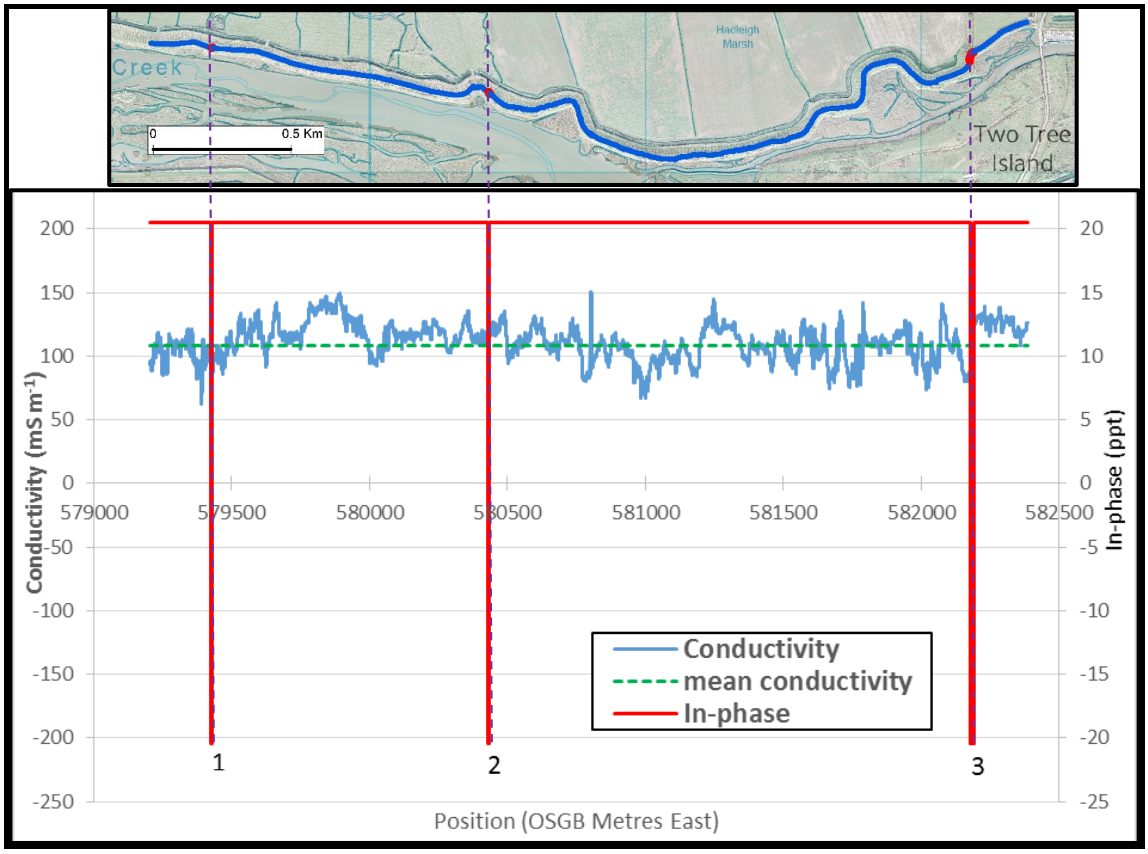



**Figure 3.9: Top: Map showing the extent of the EM31 survey along the embankment in blue (Contains Ordnance Survey data © Crown copyright and database right 2014, © GeoPerspectives). Bottom: EM31 survey results along embankment. (North up the page)**

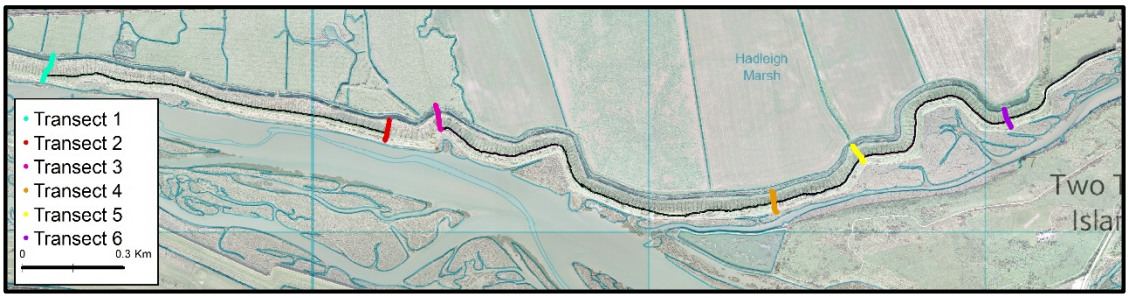



**Figure 3.10a: EM31 transect locations (Contains Ordnance Survey data © Crown copyright and database right 2014, © GeoPerspectives). (North up the page)**

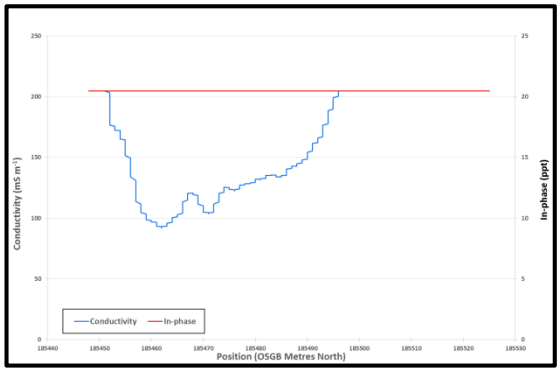



**Figure 3.10b: Transect 1**

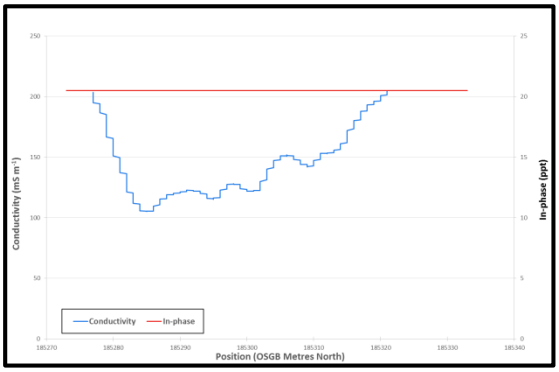



**Figure 3.10c: Transect 2**

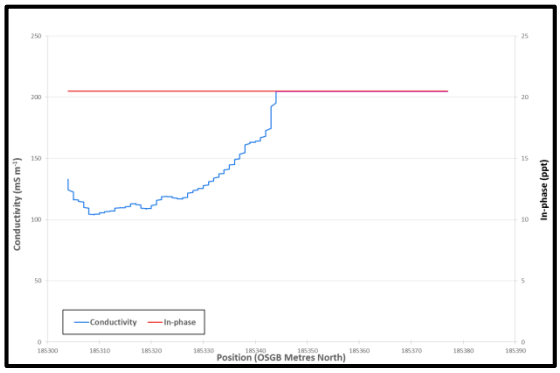



**Figure 3.10d: Transect 3**

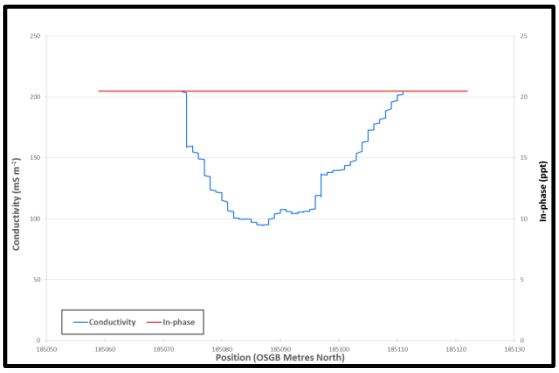



**Figure 3.10e: Transect 4**

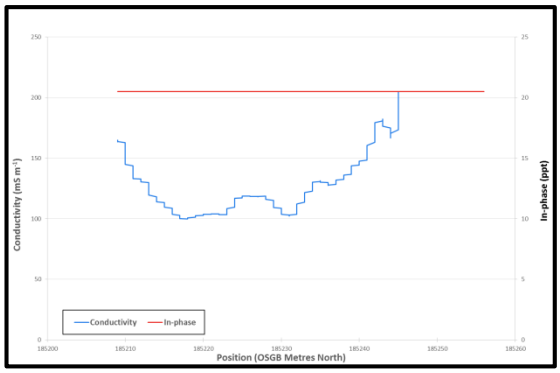



**Figure 3.10f: Transect 5**

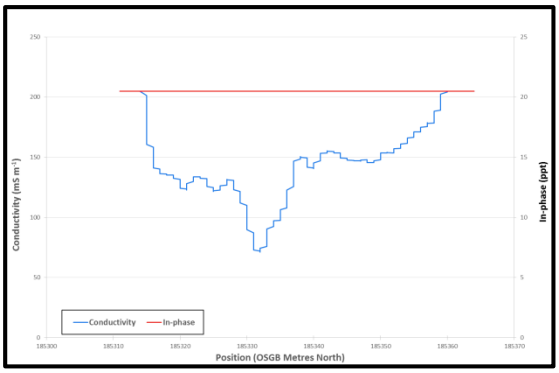



**Figure 3.10g: Transect 6**

**Figure 3.10: EM31 transverse survey results**

## 3.7 Composition of excavated waste samples

The waste extracted from Leigh Marshes landfill, trial pits 5 and 6, was predominantly composed of a brown and black, fine-grained particulate matrix interspersed with broken bricks, glass, ceramics, and small quantities of paper, rubber, bones, plant materials and wood (Plate 3.4). It was interesting to note that many bottles have survived being landfilled and some may still contain their original, potentially hazardous, contents (Plate 3.4).

The waste extracted from Hadleigh Marsh landfill, trial pits 3 and 4, had the appearance of present-day waste (before separation for recycling), consisting of plastics, ceramics, textiles (shoes, carpets, clothes), paper, wood, batteries, soil (predominantly clay), and putrescible materials (Plate 3.5).

Table 3.13 shows the moisture content of each material dried for inorganic contaminant analysis by trial pit. Statistical analyses ( Mann-Whitney U tests) were carried out to determine whether there were significant differences in moisture content between trial pits or landfill sites (see Table 3.13 for numbers of samples and median values). There was a significant difference between the moisture contents of the matrix material samples in the two Leigh Marshes trial pits ($p=0.033$), with trial pit 6 having the greater median value. There were no significant differences between the moisture contents of the matrix material samples in the two Hadleigh Marsh trial pits ($p=0.068$) or the moisture contents of the wood samples in the two Hadleigh Marsh trial pits ($p=0.070$). There were significant differences in moisture content of the paper samples between the Hadleigh Marsh trial pits ($p=0.019$), with trial pit 4 having the greater median value, and significant differences in moisture content of the textile samples between the Hadleigh Marsh trial pits ($p=0.024$), with trial pit 4 having the greater median value. Comparing the matrix material moisture content between sites, Leigh Marshes landfill has a significantly higher moisture content than Hadleigh Marsh landfill (Mann-Whitney U, Hadleigh Marsh n=24 median = 45%, Leigh Marshes n=24 median = 76%, $p<0.001$).

**Plate 3.4: Waste extracted from Leigh Marshes trial pit 6 (J. Brand, 26/3/2014) included a sealed poison bottle containing an unidentified liquid (J. Brand, 2/9/2014)**

**Plate 3.5: Waste in Hadleigh Marsh flood embankment, trial pit 3 (J. Brand, 27/3/2014)**

**Table 3.13: Moisture content in landfill waste dried for inorganic contaminant analysis calculated on a dry mass basis**

| Trial pit | Material type | Fraction | N | Min (%) | Max (%) | Range (%) | Median (%) |
|---|---|---|---|---|---|---|---|
| **Hadleigh Marsh** | | | | | | | |
| 3 | Matrix | <2 mm | 12 | 39 | 58 | 19 | 46 |
| 3 | Wood | <10 mm | 12 | 137 | 330 | 193 | 212 |
| 3 | Paper | <10 mm | 9 | 138 | 277 | 139 | 208 |
| 3 | Textiles | <10 mm | 3 | 65 | 85 | 21 | 79 |
| 4 | Matrix | <2 mm | 12 | 36 | 193 | 157 | 39 |
| 4 | Wood | <10 mm | 3 | 102 | 196 | 94 | 140 |
| 4 | Paper | <10 mm | 9 | 193 | 312 | 118 | 261 |
| 4 | Textiles | <10 mm | 6 | 96 | 215 | 118 | 121 |
| **Leigh Marshes** | | | | | | | |
| 5 | Matrix | <2 mm | 12 | 63 | 79 | 15 | 71 |
| 6 | Matrix | <2 mm | 12 | 55 | 116 | 62 | 94 |

## 3.8 Inorganic concentrations in the waste samples

### 3.8.1 Inorganic contaminant concentrations in the matrix material

#### *Leigh Marshes*

**Local variability of inorganic contaminant concentrations in the matrix material**

Metal concentrations in the <2 mm fraction of the Leigh Marshes landfill matrix material samples determined by ICP-OES and ICP-MS analyses are shown in Figure 3.11 split by trial pit. The metal concentrations in the matrix material samples from the two Leigh Marshes trial pits were highly heterogeneous. Metal concentrations in trial pit 5 vary by up to an order of magnitude between the minimum and maximum concentrations of each of the trace metals, but there is much less variation (less than one order of magnitude) between the minimum and maximum concentrations of each of the major metals. There were also differences of up to three orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 5. Metal concentrations in trial pit 6 vary by up to two orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to three orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 6.

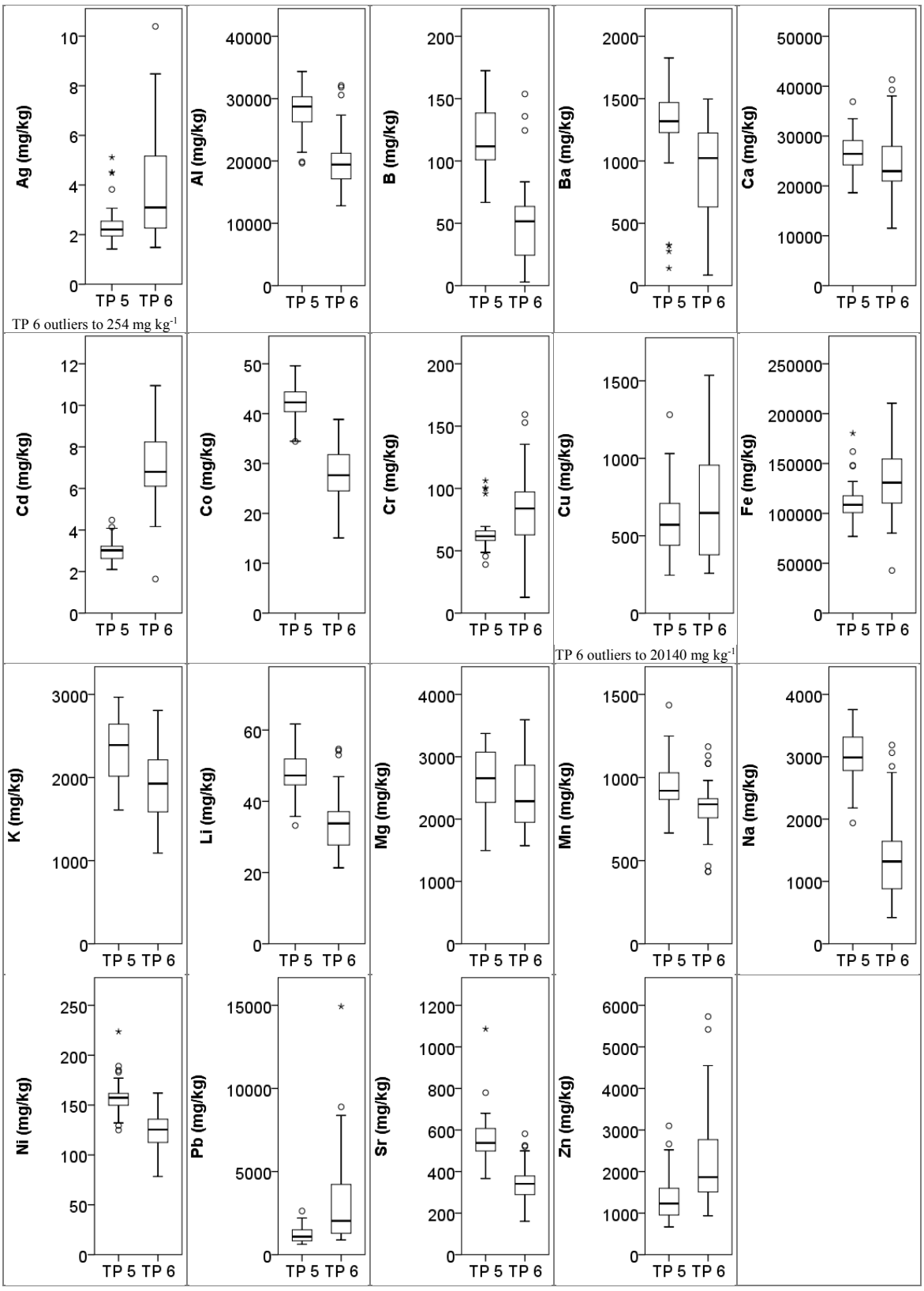



**Figure 3.11: Inorganic contaminant concentrations in the matrix material from the Leigh Marshes trial pits (trial pit 5, n=36; trial pit 6, n=35)**

### Variability of inorganic contaminant concentrations in Leigh Marshes matrix material

To test whether there were differences in contaminant concentrations in the matrix material between the two trial pits within Leigh Marshes, statistical comparisons (Mann-Whitney U tests) were carried out. Table 3.14 summarises the results of these comparisons for Leigh Marshes. Only Cu and Mg were not significantly different between the two trial pits. Trial pit 5 had significantly higher concentrations of Al, B, Ba, Ca, Co, K, Li, Mn, Na, Ni and Sr compared to trial pit 6. Trial pit 6 had significantly higher concentrations of Ag, Cd, Cr, Fe, Pb and Zn compared to trial pit 5.

**Table 3.14: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in inorganic contaminant concentrations in matrix material between the two trial pit locations within Leigh Marshes landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Trial pit 5 median (mg $kg^{-1}$) (n=36) | Trial pit 6 median (mg $kg^{-1}$) (n=35) |
|---|---|---|---|---|
| Ag | 0.002 | TP5<TP6 | 2.20 | 3.09 |
| Al | <0.001 | TP5>TP6 | 28740 | 19419 |
| B | <0.001 | TP5>TP6 | 112 | 52 |
| Ba | <0.001 | TP5>TP6 | 1319 | 1023 |
| Ca | 0.031 | TP5>TP6 | 26427 | 22967 |
| Cd | <0.001 | TP5<TP6 | 3.02 | 6.80 |
| Co | <0.001 | TP5>TP6 | 42 | 28 |
| Cr | 0.001 | TP5<TP6 | 62 | 84 |
| Cu | 0.527 | No | 571 | 647 |
| Fe | 0.008 | TP5<TP6 | 108679 | 130793 |
| K | <0.001 | TP5>TP6 | 2391 | 1927 |
| Li | <0.001 | TP5>TP6 | 47 | 34 |
| Mg | 0.126 | No | 2655 | 2287 |
| Mn | 0.001 | TP5>TP6 | 920 | 839 |
| Na | <0.001 | TP5>TP6 | 2989 | 1321 |
| Ni | <0.001 | TP5>TP6 | 157 | 125 |
| Pb | <0.001 | TP5<TP6 | 1083 | 2031 |
| Sr | <0.001 | TP5>TP6 | 538 | 341 |
| Zn | <0.001 | TP5<TP6 | 1232 | 1866 |

To investigate further the heterogeneity of the waste within Leigh Marshes landfill, the contaminants that were assessed in the matrix material both in this research and

Halcrow's previous study in 2012 are compared in Table 3.15. Statistical analysis (Mann-Whitney U tests) of the datasets for Leigh Marshes showed there to be a significant difference in measured concentrations of Cd, Cr, Cu, Pb, Ni and Zn between the two studies (all $p<0.001$), with this research finding higher concentrations of the contaminants, with the exception of Pb for which the concentrations found by Halcrow were higher.

**Table 3.15: Leigh Marshes inorganic contaminant concentrations found in the matrix material in this research vs. Halcrow study**

| | This research (n=71) | | | Halcrow study (n=30) | | |
|---|---|---|---|---|---|---|
| **Element** | **Min (mg kg$^{-1}$)** | **Max (mg kg$^{-1}$)** | **Median (mg kg$^{-1}$)** | **Min (mg kg$^{-1}$)** | **Max (mg kg$^{-1}$)** | **Median (mg kg-$^{1}$)** |
| Cd | 1.6 | 11 | 4.1 | <0.5 | 19.0 | 1.0 |
| Cr | 13 | 159 | 64 | 20 | 75 | 35 |
| Cu | 245 | 20140 | 605 | 22 | 1191 | 145 |
| Ni | 78 | 224 | 141 | 16 | 177 | 39 |
| Pb | 625 | 14936 | 1332 | 34 | 3564 | 555 |
| Zn | 667 | 5731 | 1540 | 72 | 1578 | 527 |

Table 3.16 shows the metal concentrations in the matrix material for the overall site, i.e. data for trial pits 5 and 6 combined. The metal concentrations were highly heterogeneous, differing by up to two orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to three orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in the Leigh Marshes matrix material.

**Table 3.16: Inorganic contaminant concentrations in <2 mm fraction of matrix material waste samples from Leigh Marshes (n=71)**

| Element | Limit of Detection (mg $kg^{-1}$) | Minimum (mg $kg^{-1}$) | Maximum (mg $kg^{-1}$) | Range (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
|---|---|---|---|---|---|
| Ag | 0.06 | 1.42 | 254 | 253 | 2.40 |
| Al | 0.12 | 12815 | 34355 | 21540 | 25284 |
| B | 5.83 | <LOD | 172 | 169 | 85 |
| Ba | 0.09 | 84 | 1827 | 1743 | 1232 |
| Ca | 0.16 | 11507 | 41292 | 29785 | 24553 |
| Cd | 0.72* | 1.64 | 11 | 9.4 | 4.14 |
| Co | 0.31 | 15 | 50 | 35 | 38 |
| Cr | 0.12 | 13 | 159 | 146 | 64 |
| Cu | 0.08 | 245 | 20140 | 19895 | 605 |
| Fe | 6.55 | 42710 | 210530 | 167820 | 114813 |
| K | 0.17 | 1091 | 2967 | 1876 | 2177 |
| Li | 0.03 | 21 | 62 | 41 | 44 |
| Mg | 0.09 | 1495 | 3594 | 2099 | 2564 |
| Mn | 0.07 | 433 | 1436 | 1003 | 870 |
| Na | 0.07 | 420 | 3756 | 3336 | 2588 |
| Ni | 0.60 | 78 | 224 | 146 | 141 |
| Pb | 1.11 | 625 | 14936 | 14311 | 1332 |
| Sr | 0.05 | 161 | 1086 | 925 | 472 |
| Zn | 0.17 | 667 | 5731 | 5064 | 1540 |
| *Trial pit 5 samples analysed using ICP-MS, LOD = 0.005 mg $kg^{-1}$ | | | | | |

## Correlation analyses

Spearman's rank correlation coefficients are shown in Table 3.17 for tests to determine whether there were correlations between metal concentrations or between metal concentrations and moisture content in Leigh Marshes matrix material. Strong correlations between metals could indicate they are from a common source, and correlations between metals and moisture content could indicate that the metal concentrations have been influenced by leachate movement. Zinc showed a weak positive correlation with moisture content, and Ag, Cd, Cr and Fe showed moderate positive correlations with moisture content. Calcium and Mn showed weak negative correlations with moisture content, and Al, Co, K, Li, Mg, Na, Ni and Sr showed

moderate negative correlations with moisture content. There were many correlations between the metals, both positive and negative, but few strong correlations were found.

**Table 3.17: Spearman's rank correlation coefficients for metals and moisture content (MC) in Leigh Marshes matrix material. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8**

| | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **MC** | 0.54 | -0.70 | | | -0.50 | 0.55 | -0.57 | 0.69 | | 0.57 | -0.60 | -0.72 | -0.58 | -0.50 | -0.65 | -0.53 | | -0.55 | 0.47 |
| **Ag** | | -0.38 | -0.32 | -0.29 | | 0.40 | -0.36 | 0.53 | | 0.34 | -0.32 | -0.41 | -0.28 | | -0.43 | | 0.44 | -0.42 | 0.39 |
| **Al** | | | 0.54 | 0.37 | 0.68 | -0.57 | **0.80** | -0.41 | -0.25 | -0.25 | **0.84** | **0.95** | 0.65 | 0.67 | **0.90** | 0.70 | -0.47 | **0.84** | -0.42 |
| **B** | | | | 0.44 | 0.24 | -0.47 | 0.65 | | | | 0.34 | 0.54 | | 0.41 | 0.59 | 0.62 | -0.52 | 0.57 | -0.40 |
| **Ba** | | | | | | -0.36 | 0.38 | | | | 0.27 | 0.29 | | | 0.35 | 0.31 | -0.42 | 0.42 | |
| **Ca** | | | | | | -0.27 | 0.55 | | | | 0.53 | 0.71 | 0.62 | 0.75 | 0.66 | 0.39 | | 0.62 | |
| **Cd** | | | | | | | -0.71 | 0.55 | | 0.50 | -0.44 | -0.57 | -0.28 | -0.32 | -0.68 | -0.58 | 0.51 | -0.69 | 0.58 |
| **Co** | | | | | | | | -0.35 | | | 0.56 | **0.81** | 0.39 | 0.69 | **0.85** | **0.88** | -0.48 | **0.86** | -0.54 |
| **Cr** | | | | | | | | | 0.36 | 0.52 | -0.39 | -0.44 | -0.28 | | -0.49 | | 0.52 | -0.40 | 0.44 |
| **Cu** | | | | | | | | | | | | | | | | | | | 0.24 |
| **Fe** | | | | | | | | | | | -0.46 | -0.30 | -0.42 | | -0.36 | | 0.25 | | |
| **K** | | | | | | | | | | | | **0.82** | 0.78 | 0.44 | 0.75 | 0.50 | -0.37 | 0.59 | |
| **Li** | | | | | | | | | | | | | 0.71 | 0.72 | **0.92** | 0.69 | -0.46 | **0.83** | -0.40 |
| **Mg** | | | | | | | | | | | | | | 0.57 | 0.62 | 0.32 | | 0.43 | |
| **Mn** | | | | | | | | | | | | | | | 0.65 | 0.59 | | 0.70 | |
| **Na** | | | | | | | | | | | | | | | | 0.71 | -0.49 | **0.88** | -0.40 |
| **Ni** | | | | | | | | | | | | | | | | | -0.34 | 0.71 | -0.46 |
| **Pb** | | | | | | | | | | | | | | | | | | -0.44 | 0.48 |
| **Sr** | | | | | | | | | | | | | | | | | | | -0.44 |
| **Zn** | | | | | | | | | | | | | | | | | | | |

## *Hadleigh Marsh*

### Local variability of inorganic contaminant concentrations in the matrix material

Metal concentrations in the <2 mm fraction of the Hadleigh Marsh landfill matrix material samples determined by ICP-OES and ICP-MS analysis are shown in Figure 3.12 split by trial pit. The metal concentrations in the matrix material samples from the two Hadleigh Marsh trial pits were highly heterogeneous. Metal concentrations in trial pit 3 vary by up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also

differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 3. Metal concentrations in trial pit 4 vary by up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 4.

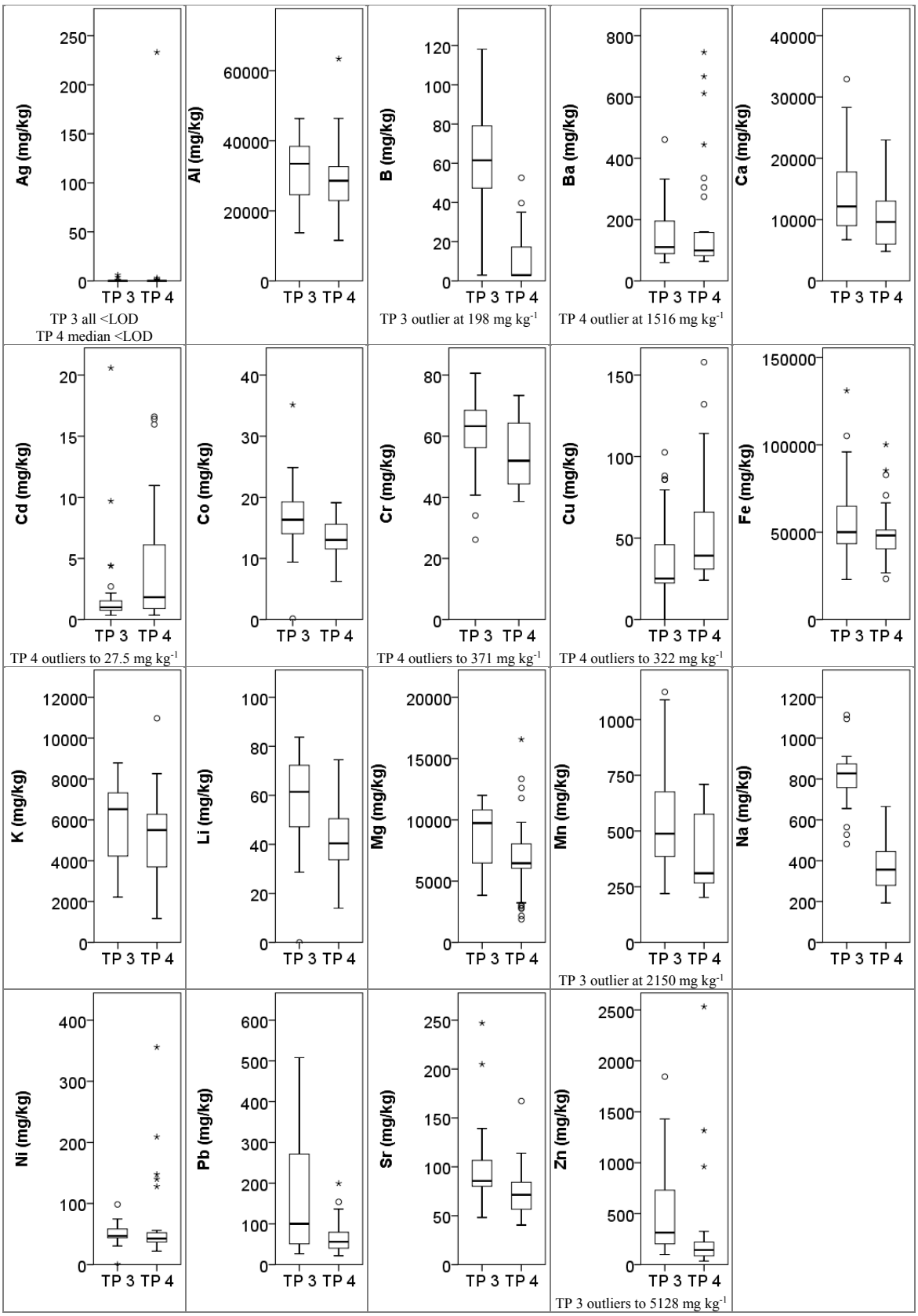



**Figure 3.12: Inorganic contaminant concentrations in the matrix material from the Hadleigh Marsh trial pits (trial pit 3, n=35; trial pit 4, n=36)**

## Variability of inorganic contaminant concentrations in Hadleigh Marsh matrix material

To test whether there were differences in contaminant concentrations in the matrix material between the two trial pits within Hadleigh Marsh, statistical comparisons (Mann-Whitney U tests) were carried out. Table 3.18 summarises the results of these comparisons for Hadleigh Marsh, only Al, Ba, Fe and Ni were not significantly different between the two trial pits. Trial pit 3 had significantly higher concentrations of B, Ca, Co, Cr, K, Li, Mg, Mn, Na, Pb, Sr and Zn compared to trial pit 4. Trial pit 4 had significantly higher concentrations of Cd and Cu compared to trial pit 3.

**Table 3.18: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in inorganic contaminant concentrations in matrix material between the two trial pit locations within Hadleigh Marsh landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Trial pit 3 median (mg $kg^{-1}$) (n=35) | Trial pit 4 median (mg $kg^{-1}$) (n=36) |
|---|---|---|---|---|
| Ag | Insufficient data | | <LOD | <LOD |
| Al | 0.064 | No | 33454 | 28600 |
| B | <0.001 | TP3>TP4 | 61 | <LOD |
| Ba | 0.818 | No | 110 | 99 |
| Ca | 0.003 | TP3>TP4 | 12145 | 9616 |
| Cd | 0.017 | TP3<TP4 | 1.00 | 1.83 |
| Co | 0.002 | TP3>TP4 | 16 | 13 |
| Cr | 0.012 | TP3>TP4 | 63 | 52 |
| Cu | 0.001 | TP3<TP4 | 25 | 39 |
| Fe | 0.135 | No | 50068 | 48157 |
| K | 0.028 | TP3>TP4 | 6520 | 5502 |
| Li | <0.001 | TP3>TP4 | 61 | 40 |
| Mg | 0.005 | TP3>TP4 | 9735 | 6456 |
| Mn | 0.004 | TP3>TP4 | 488 | 310 |
| Na | <0.001 | TP3>TP4 | 827 | 356 |
| Ni | 0.151 | No | 47 | 43 |
| Pb | 0.004 | TP3>TP4 | 100 | 56 |
| Sr | <0.001 | TP3>TP4 | 86 | 71 |
| Zn | <0.001 | TP3>TP4 | 314 | 144 |

To investigate further the heterogeneity of the waste within Hadleigh Marsh landfill, the contaminants that were assessed in the matrix material both in this research and

Halcrow's previous study in 2012 are compared in Table 3.19. Statistical analysis (Mann-Whitney U tests) of the datasets for Hadleigh Marsh showed there to be a significant difference in measured concentrations of Cd ($p<0.001$), Cr ($p<0.001$), Ni ($p<0.001$) and Zn ($p=0.010$) between the two studies, with this research finding higher concentrations of the four contaminants. There were no significant differences between the two studies for Cu ($p=0.097$) and Pb ($p=0.183$) concentrations in the Hadleigh Marsh samples.

**Table 3.19: Hadleigh Marsh inorganic contaminant concentrations found in the matrix material in this research vs. Halcrow study**

| | This research (n=71) | | | Halcrow study (n=25) | | |
|---|---|---|---|---|---|---|
| **Element** | **Min (mg kg$^{-1}$)** | **Max (mg kg$^{-1}$)** | **Median (mg kg$^{-1}$)** | **Min (mg kg$^{-1}$)** | **Max (mg kg$^{-1}$)** | **Median (mg kg-$^{1}$)** |
| Cd | 0.3 | 28 | 1.2 | <0.5 | 1.5 | <0.5 |
| Cr | 26 | 371 | 59 | 8 | 86 | 27 |
| Cu | <0.08 | 322 | 33 | 13 | 232 | 30 |
| Ni | <0.06 | 356 | 45 | 10 | 31 | 20 |
| Pb | 22 | 508 | 63 | 27 | 5340 | 106 |
| Zn | 35 | 5128 | 209 | 53 | 41257 | 125 |

Table 3.20 shows the metal concentrations in the matrix material for the overall site, i.e. data for trial pits 3 and 4 combined. The metal concentrations were highly heterogeneous, differing by up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals.

**Table 3.20: Inorganic contaminant concentrations in <2 mm fraction of matrix material waste samples from Hadleigh Marsh (n=71)**

| Element | Limit of Detection (mg $kg^{-1}$) | Minimum (mg $kg^{-1}$) | Maximum (mg $kg^{-1}$) | Range (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
|---|---|---|---|---|---|
| Ag | 0.06 | <LOD | 233 | 233 | <LOD |
| Al | 0.12 | 11575 | 63428 | 51853 | 30428 |
| B | 5.83 | <LOD | 198 | 192 | 30 |
| Ba | 0.09 | 60 | 1516 | 1456 | 103 |
| Ca | 0.16 | 4813 | 32920 | 28107 | 11116 |
| Cd | 0.005 | 0.35 | 28 | 27.65 | 1.18 |
| Co | 0.31 | <LOD | 35 | 35 | 15 |
| Cr | 0.12 | 26 | 371 | 345 | 59 |
| Cu | 0.08 | <LOD | 322 | 322 | 33 |
| Fe | 6.55 | 22971 | 131132 | 108161 | 48533 |
| K | 0.17 | 1171 | 10971 | 9800 | 5713 |
| Li | 0.03 | <LOD | 84 | 84 | 49 |
| Mg | 0.09 | 1886 | 16567 | 14681 | 7924 |
| Mn | 0.07 | 202 | 2510 | 2308 | 422 |
| Na | 0.07 | 193 | 1113 | 920 | 559 |
| Ni | 0.60 | <LOD | 356 | 356 | 45 |
| Pb | 1.11 | 22 | 508 | 486 | 63 |
| Sr | 0.05 | 40 | 247 | 207 | 81 |
| Zn | 0.17 | 35 | 5128 | 5093 | 209 |

## Correlation analyses

Spearman's rank correlation coefficients are shown in Table 3.21 for tests to determine whether there were correlations between metal concentrations or between metal concentrations and moisture content in Hadleigh Marsh matrix material. Strong correlations between metals could indicate they are from a common source, and correlations between metals and moisture content could indicate that the metal concentrations have been influenced by leachate movement. Cadmium showed a weak positive correlation with moisture content, and Na, Ni, Pb and Zn showed a moderate positive correlation with moisture content. There were many correlations between the metals, both positive and negative, but very few strong correlations were found.

**Table 3.21: Spearman's rank correlation coefficients for metals and moisture content (MC) in Hadleigh Marsh matrix material. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8. Ag omitted as median value<LOD**

| | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **MC** | | | | | 0.50 | | | | | | | | | 0.62 | 0.51 | 0.75 | | 0.73 |
| **Al** | | 0.39 | -0.35 | | -0.58 | 0.54 | 0.61 | -0.40 | | **0.97** | **0.86** | **0.92** | | | 0.24 | -0.48 | | -0.25 |
| **B** | | | | 0.49 | -0.25 | 0.37 | 0.47 | | 0.30 | 0.35 | 0.53 | 0.42 | 0.41 | 0.79 | 0.30 | 0.23 | 0.58 | 0.39 |
| **Ba** | | | | | 0.33 | | | 0.39 | | -0.37 | -0.28 | -0.39 | | | 0.29 | 0.58 | 0.45 | 0.55 |
| **Ca** | | | | | | | | | 0.30 | | | | | 0.54 | 0.24 | 0.43 | 0.75 | 0.52 |
| **Cd** | | | | | | -0.30 | | 0.61 | 0.27 | -0.53 | -0.63 | -0.66 | | | | 0.48 | | 0.42 |
| **Co** | | | | | | | 0.67 | | 0.35 | 0.54 | 0.66 | 0.57 | 0.58 | 0.28 | 0.67 | | 0.26 | |
| **Cr** | | | | | | | | | 0.42 | 0.60 | 0.62 | 0.53 | 0.43 | 0.35 | 0.66 | | 0.30 | |
| **Cu** | | | | | | | | | 0.32 | -0.40 | -0.49 | -0.49 | | | 0.30 | 0.46 | | 0.34 |
| **Fe** | | | | | | | | | | | | | 0.60 | | 0.66 | 0.27 | 0.39 | 0.37 |
| **K** | | | | | | | | | | | **0.85** | **0.92** | | | 0.24 | -0.45 | | |
| **Li** | | | | | | | | | | | | **0.90** | 0.35 | 0.41 | 0.25 | -0.35 | | |
| **Mg** | | | | | | | | | | | | | 0.23 | | | -0.43 | | |
| **Mn** | | | | | | | | | | | | | | 0.34 | 0.47 | | 0.28 | |
| **Na** | | | | | | | | | | | | | | | 0.24 | 0.52 | 0.60 | 0.68 |
| **Ni** | | | | | | | | | | | | | | | | | 0.44 | 0.26 |
| **Pb** | | | | | | | | | | | | | | | | | 0.47 | **0.83** |
| **Sr** | | | | | | | | | | | | | | | | | | 0.58 |
| **Zn** | | | | | | | | | | | | | | | | | | |

## *Inorganic contaminant variability between sites*

The statistical comparison (Mann-Whitney U tests) of metal concentrations in Leigh Marshes and Hadleigh Marsh landfills is summarised in Table 3.22. Leigh Marshes landfill contains significantly higher concentrations of Ag, B, Ba, Ca, Cd, Co, Cr, Cu, Fe, Mn, Na, Ni, Pb, Sr and Zn in the matrix material than Hadleigh Marsh landfill. Hadleigh Marsh landfill contains significantly higher concentrations of Al, K, Li and Mg in the matrix material than Leigh Marshes landfill.

**Table 3.22: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between inorganic contaminant concentrations in the matrix material at Leigh Marshes (LM) landfill and Hadleigh Marsh (HM) landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (mg $kg^{-1}$) (n=71) | Hadleigh Marsh median (mg $kg^{-1}$) (n=71) |
|---|---|---|---|---|
| Ag | <0.001 | LM>HM | 2.40 | <LOD |
| Al | <0.001 | LM<HM | 25284 | 30428 |
| B | <0.001 | LM>HM | 85 | 30 |
| Ba | <0.001 | LM>HM | 1232 | 103 |
| Ca | <0.001 | LM>HM | 24553 | 11116 |
| Cd | <0.001 | LM>HM | 4.14 | 1.18 |
| Co | <0.001 | LM>HM | 38 | 15 |
| Cr | <0.001 | LM>HM | 64 | 59 |
| Cu | <0.001 | LM>HM | 605 | 33 |
| Fe | <0.001 | LM>HM | 114813 | 48533 |
| K | <0.001 | LM<HM | 2177 | 5713 |
| Li | 0.006 | LM<HM | 44 | 49 |
| Mg | <0.001 | LM<HM | 2564 | 7924 |
| Mn | <0.001 | LM>HM | 870 | 422 |
| Na | <0.001 | LM>HM | 2588 | 559 |
| Ni | <0.001 | LM>HM | 141 | 45 |
| Pb | <0.001 | LM>HM | 1332 | 63 |
| Sr | <0.001 | LM>HM | 472 | 81 |
| Zn | <0.001 | LM>HM | 1540 | 209 |

## 3.8.2 Inorganic contaminant concentrations in wood

### Local variability of inorganic contaminant concentrations in the wood samples

Metal concentrations in the Hadleigh Marsh landfill wood samples determined by ICP-OES analysis are shown in Figure 3.13 split by trial pit. The metal concentrations in the wood samples from the two trial pits were highly heterogeneous. Metal concentrations in trial pit 3 vary by up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to five orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 3.

Metal concentrations in trial pit 4 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 4.

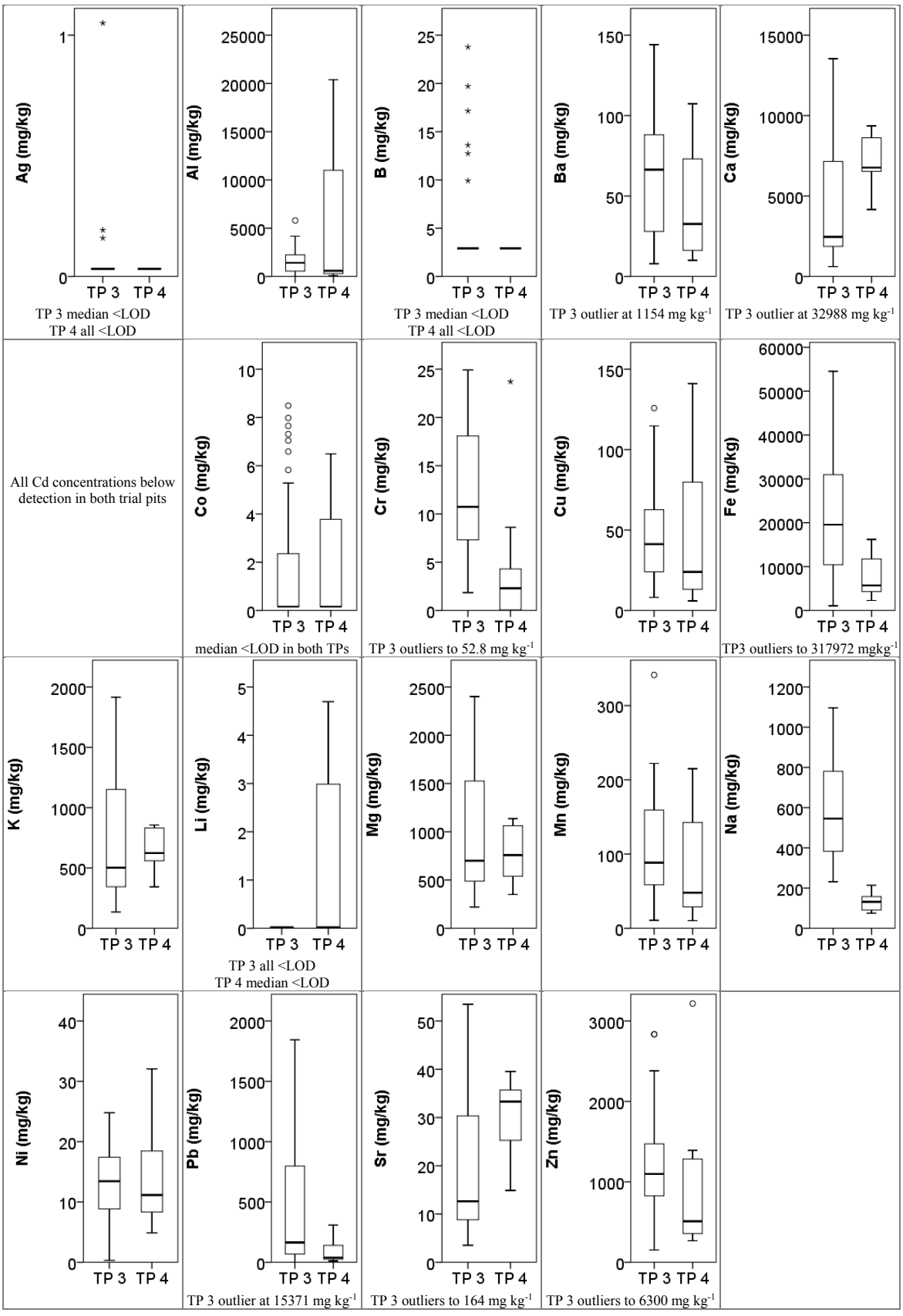



**Figure 3.13: Inorganic contaminant concentrations in the wood samples from the Hadleigh Marsh trial pits (trial pit 3, n=36; trial pit 4, n=9)**

### Variability of inorganic contaminant concentrations in Hadleigh Marsh wood

To test whether there were differences in contaminant concentrations in the wood samples between the two trial pits within Hadleigh Marsh, statistical comparisons (Mann-Whitney U tests) were carried out. The following metals were not included in the statistical analysis as more than 50% of the data were below the LOD: Ag, B, Cd, Co and Li. Table 3.23 summarises the results of these comparisons, Al, Ba, Cu, K, Mg, Mn, Ni and Zn were not significantly different between the two trial pits. Trial pit 3 had significantly higher concentrations of Cr, Fe, Na and Pb in the wood samples compared to trial pit 4. Trial pit 4 had significantly higher concentrations of Ca and Sr in the wood samples compared to trial pit 3.

**Table 3.23: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in inorganic contaminant concentrations in wood samples between the two trial pit locations within Hadleigh Marsh landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Trial pit 3 median (mg $kg^{-1}$) (n=36) | Trial pit 4 median (mg $kg^{-1}$) (n=9) |
|---|---|---|---|---|
| Ag | insufficient data | | <LOD | <LOD |
| Al | 0.551 | No | 1414 | 586 |
| B | insufficient data | | <LOD | <LOD |
| Ba | 0.294 | No | 66.3 | 32.6 |
| Ca | 0.027 | TP3<TP4 | 2461 | 6767 |
| Cd | insufficient data | | <LOD | <LOD |
| Co | insufficient data | | <LOD | <LOD |
| Cr | 0.002 | TP3>TP4 | 10.7 | 2.291 |
| Cu | 0.294 | No | 41.2 | 24.0 |
| Fe | 0.006 | TP3>TP4 | 19549 | 5698 |
| K | 0.478 | No | 502 | 624 |
| Li | insufficient data | | <LOD | <LOD |
| Mg | 0.777 | No | 700 | 758 |
| Mn | 0.212 | No | 88.3 | 48.0 |
| Na | <0.001 | TP3>TP4 | 546 | 131 |
| Ni | 0.820 | No | 13.4 | 11.1 |
| Pb | 0.010 | TP3>TP4 | 164 | 38.9 |
| Sr | 0.036 | TP3<TP4 | 12.6 | 33.3 |
| Zn | 0.156 | No | 1098 | 510 |

Table 3.24 shows the metal concentrations in the wood samples for the overall site, i.e. data for trial pits 3 and 4 combined. Metal concentrations in the wood were highly heterogeneous, differing by up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to five orders of magnitude between the median concentrations of different trace metals and differences of up to an order of magnitude between the median concentrations of different major metals.

**Table 3.24: Inorganic contaminant concentrations in <10 mm fraction of wood waste samples from Hadleigh Marsh (n=45)**

| Element | Limit of Detection (mg $kg^{-1}$) | Minimum (mg $kg^{-1}$) | Maximum (mg $kg^{-1}$) | Range (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
|---|---|---|---|---|---|
| Ag | 0.06 | <LOD | 1.05 | 1 | <LOD |
| Al | 0.12 | 13 | 20390 | 20377 | 1299 |
| B | 5.83 | <LOD | 24 | 21 | <LOD |
| Ba | 0.09 | 7.92 | 1154 | 1146 | 42 |
| Ca | 0.16 | 613 | 32988 | 32375 | 3880 |
| Cd | 0.72 | <LOD | <LOD | - | <LOD |
| Co | 0.31 | <LOD | 8.49 | 8 | <LOD |
| Cr | 0.12 | 0.06 | 53 | 53 | 10 |
| Cu | 0.08 | 5.91 | 141 | 135 | 36 |
| Fe | 6.55 | 1057 | 317972 | 316915 | 13096 |
| K | 0.17 | 135 | 1915 | 1780 | 535 |
| Li | 0.03 | <LOD | 4.70 | 4.7 | <LOD |
| Mg | 0.09 | 220 | 2401 | 2181 | 703 |
| Mn | 0.07 | 10 | 341 | 331 | 84 |
| Na | 0.07 | 75 | 1096 | 1021 | 451 |
| Ni | 0.60 | <LOD | 32 | 32 | 13 |
| Pb | 1.11 | <LOD | 15271 | 15270 | 140 |
| Sr | 0.05 | 3.53 | 164 | 160 | 18 |
| Zn | 0.17 | 153 | 6300 | 6147 | 1094 |

## Correlation analyses

Spearman's rank correlation coefficients are shown in Table 3.25 for tests to determine whether there were correlations between metal concentrations or between metal

concentrations and moisture content in the Hadleigh Marsh wood samples. Strong correlations between metals could indicate they are from a common source, and correlations between metals and moisture content could indicate that the metal concentrations have been influenced by leachate movement. Potassium, Mg and Na showed moderate positive correlations with moisture content. There were no negative correlations between the metals in the wood samples. There were many positive correlations between the metals, but few strong correlations were found.

**Table 3.25: Spearman's rank correlation coefficients for metals and moisture content (MC) in Hadleigh Marsh wood samples. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show rs>0.8. Ag, B, Cd, Co and Li omitted as median values<LOD**

| | Al | Ba | Ca | Cr | Cu | Fe | K | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **MC** | | | | | | | 0.52 | 0.65 | | 0.67 | | | | |
| **Al** | | 0.78 | 0.41 | 0.63 | 0.66 | 0.54 | 0.68 | 0.65 | 0.79 | 0.42 | 0.77 | 0.46 | 0.53 | 0.47 |
| **Ba** | | | 0.63 | 0.50 | 0.68 | | 0.76 | 0.76 | 0.74 | 0.52 | 0.69 | 0.57 | 0.70 | 0.50 |
| **Ca** | | | | | | | **0.84** | **0.87** | 0.47 | 0.34 | 0.42 | | **0.98** | |
| **Cr** | | | | | 0.42 | 0.75 | 0.35 | 0.39 | 0.66 | 0.62 | 0.41 | | | |
| **Cu** | | | | | | | 0.41 | 0.40 | 0.60 | | 0.44 | 0.73 | 0.32 | 0.75 |
| **Fe** | | | | | | | | | 0.53 | 0.48 | 0.41 | | | |
| **K** | | | | | | | | **0.90** | 0.63 | 0.51 | 0.59 | 0.31 | **0.89** | |
| **Mg** | | | | | | | | | 0.66 | 0.67 | 0.55 | 0.31 | **0.90** | |
| **Mn** | | | | | | | | | | 0.47 | 0.63 | 0.37 | 0.57 | 0.41 |
| **Na** | | | | | | | | | | | | 0.34 | 0.36 | |
| **Ni** | | | | | | | | | | | | 0.43 | 0.50 | 0.44 |
| **Pb** | | | | | | | | | | | | | | **0.86** |
| **Sr** | | | | | | | | | | | | | | |
| **Zn** | | | | | | | | | | | | | | |

### 3.8.3 Inorganic contaminant concentrations in paper

**Local variability of inorganic contaminant concentrations in the paper samples**

Metal concentrations in the Hadleigh Marsh landfill paper samples determined by ICP-OES and ICP-MS analysis are shown in Figure 3.14 split by trial pit. The metal concentrations in the paper samples from the two trial pits were highly heterogeneous. Metal concentrations in trial pit 3 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 3. Metal concentrations in trial pit 4 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to two orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to three orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 4.

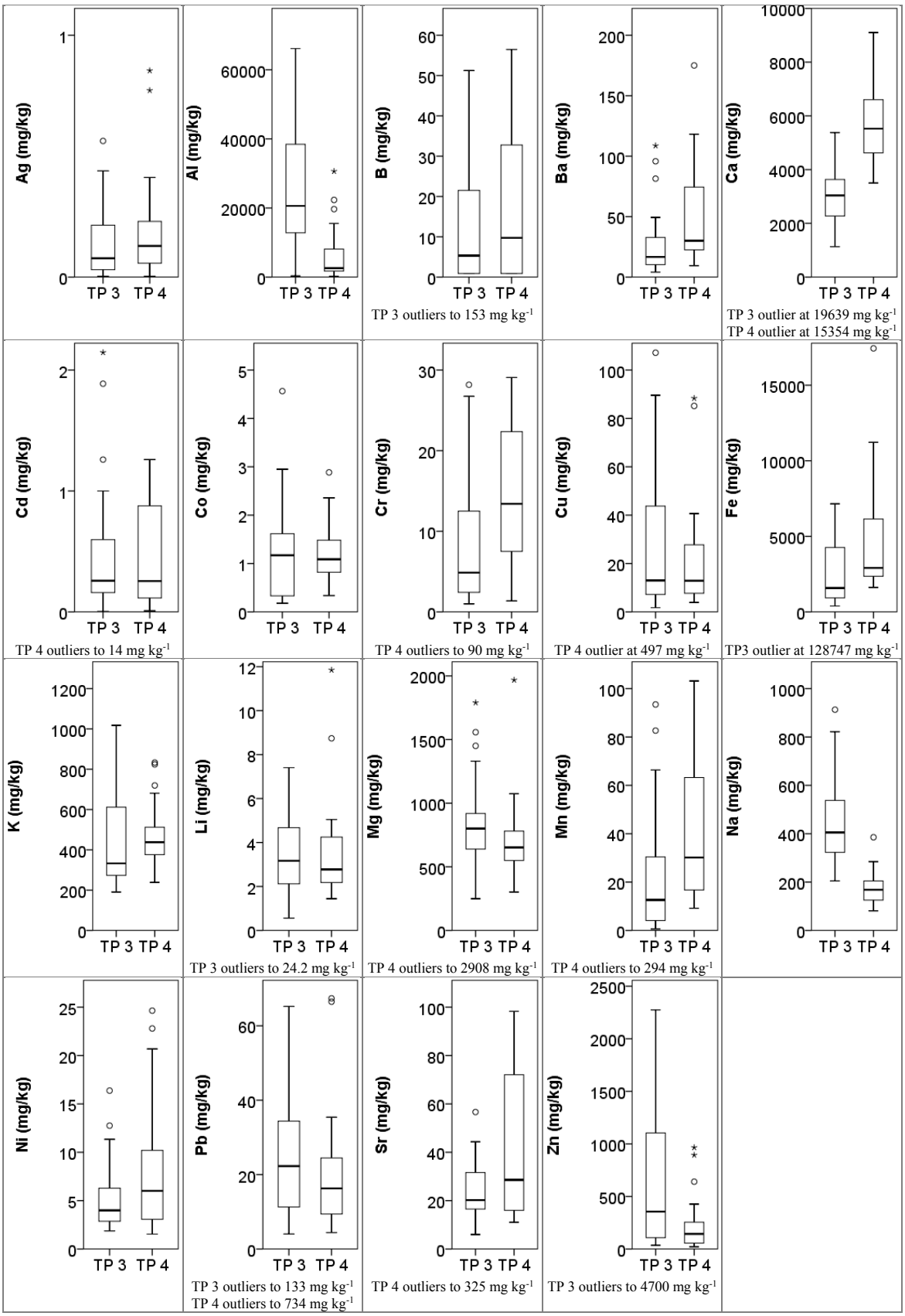



**Figure 3.14: Inorganic contaminant concentrations in the paper samples from the Hadleigh Marsh trial pits (trial pit 3, n=26; trial pit 4, n=27)**

### Variability of inorganic contaminant concentrations in Hadleigh Marsh paper

To test whether there were differences in contaminant concentrations in the paper samples between the two trial pits within Hadleigh Marsh, statistical comparisons (Mann-Whitney U tests) were carried out. Table 3.23 summarises the results of these comparisons, Ag, B, Cd, Co, Cu, K, Li, Mg, Ni and Pb were not significantly different between the two trial pits. Paper samples from trial pit 3 had significantly higher concentrations of Al, Na and Zn compared to trial pit 4. Paper samples from trial pit 4 had significantly higher concentrations of Ba, Ca, Cr, Fe, Mn and Sr compared to trial pit 3.

**Table 3.26: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in inorganic contaminant concentrations in paper samples between the two trial pit locations within Hadleigh Marsh landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Trial pit 3 median (mg $kg^{-1}$) (n=26) | Trial pit 4 median (mg $kg^{-1}$) (n=27) |
|---|---|---|---|---|
| Ag | 0.206 | No | 0.08 | 0.13 |
| Al | <0.001 | TP3>TP4 | 20607 | 2581 |
| B | 0.594 | No | 5.3 | 9.7 |
| Ba | 0.004 | TP3<TP4 | 16.7 | 30.0 |
| Ca | <0.001 | TP3<TP4 | 3041 | 5525 |
| Cd | 0.831 | No | 0.26 | 0.25 |
| Co | 0.557 | No | 1.17 | 1.09 |
| Cr | 0.006 | TP3<TP4 | 4.9 | 13.4 |
| Cu | 0.986 | No | 13.0 | 12.9 |
| Fe | 0.006 | TP3<TP4 | 1575 | 2912 |
| K | 0.117 | No | 333 | 438 |
| Li | 0.887 | No | 3.2 | 2.8 |
| Mg | 0.102 | No | 800 | 652 |
| Mn | 0.002 | TP3<TP4 | 12.6 | 30.1 |
| Na | <0.001 | TP3>TP4 | 405 | 168 |
| Ni | 0.165 | No | 4.0 | 6.0 |
| Pb | 0.233 | No | 22.3 | 16.3 |
| Sr | 0.098 | TP3<TP4 | 20.2 | 28.6 |
| Zn | 0.021 | TP3>TP4 | 356 | 144 |

Table 3.27 shows the metal concentrations in the paper samples for the overall site, i.e. data for trial pits 3 and 4 combined. Metal concentrations in the paper were highly

heterogeneous, differing by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to three orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals.

**Table 3.27: Inorganic contaminant concentrations in <10 mm fraction of paper waste samples from Hadleigh Marsh (n=53)**

| Element | Limit of Detection (mg $kg^{-1}$) | Minimum (mg $kg^{-1}$) | Maximum (mg $kg^{-1}$) | Range (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
|---|---|---|---|---|---|
| Ag | 0.006 | <LOD | 0.85 | 0.8 | 0.10 |
| Al | 6.66 | 238 | 66150 | 65912 | 8982 |
| B | 1.79 | <LOD | 153 | 152 | 7.83 |
| Ba | 0.08 | 4.05 | 175 | 171 | 24 |
| Ca | 2.62 | 1131 | 19639 | 18508 | 4589 |
| Cd | 0.005 | <LOD | 14 | 14 | 0.25 |
| Co | 0.001 | 0.18 | 4.56 | 4.4 | 1.09 |
| Cr | 0.007 | 0.99 | 90 | 89 | 9.32 |
| Cu | 1.73 | 1.73 | 497 | 495 | 13 |
| Fe | 3.81 | 390 | 128747 | 128357 | 2530 |
| K | 31 | 191 | 1018 | 827 | 414 |
| Li | 1.13 | <LOD | 24 | 23 | 3.02 |
| Mg | 0.04 | 249 | 2908 | 2659 | 702 |
| Mn | 0.22 | 0.63 | 294 | 293 | 22 |
| Na | 4.25 | 81 | 913 | 832 | 267 |
| Ni | 0.02 | 1.54 | 25 | 23 | 4.93 |
| Pb | 0.009 | 4.04 | 734 | 730 | 19 |
| Sr | 0.04 | 5.99 | 325 | 319 | 22 |
| Zn | 5.22 | 21 | 4700 | 4679 | 168 |

## Correlation analyses

Spearman's rank correlation coefficients are shown in Table 3.28 for tests to determine whether there were correlations between metal concentrations or between metal concentrations and moisture content in the Hadleigh Marsh paper samples. Strong correlations between metals could indicate they are from a common source, and

correlations between metals and moisture content could indicate that the metal concentrations have been influenced by leachate movement. Lead showed a weak negative correlation with moisture content, and Al showed a strong negative correlation with moisture content. There were many correlations between the metals, both positive and negative, but no strong correlations were found.

**Table 3.28: Spearman's rank correlation coefficients for metals and moisture content in Hadleigh Marsh paper samples. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8.**

| | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **MC** | | **-0.89** | | | | | | | | | | | | | | | -0.48 | | |
| **Ag** | | | -0.32 | 0.33 | | 0.35 | 0.48 | 0.34 | 0.55 | | 0.60 | 0.33 | | 0.47 | | 0.33 | | | |
| **Al** | | | | | -0.45 | | | | | -0.32 | | 0.34 | | -0.45 | | | 0.32 | | |
| **B** | | | | | 0.34 | | | 0.28 | | 0.36 | | | | | | 0.29 | | | |
| **Ba** | | | | | 0.55 | | 0.51 | 0.60 | 0.52 | 0.41 | 0.66 | 0.52 | | 0.47 | -0.51 | 0.55 | 0.57 | 0.58 | -0.33 |
| **Ca** | | | | | | | | 0.33 | | 0.54 | 0.44 | | | 0.66 | -0.45 | 0.50 | | 0.48 | |
| **Cd** | | | | | | | 0.34 | 0.39 | 0.58 | 0.31 | 0.55 | | 0.35 | 0.49 | | 0.44 | 0.49 | | |
| **Co** | | | | | | | | 0.51 | 0.77 | 0.68 | 0.70 | 0.59 | 0.28 | 0.65 | | 0.74 | 0.68 | | 0.42 |
| **Cr** | | | | | | | | | 0.66 | 0.60 | 0.50 | 0.35 | | 0.39 | -0.60 | 0.62 | 0.62 | | |
| **Cu** | | | | | | | | | | 0.50 | 0.77 | 0.64 | 0.38 | 0.56 | | 0.63 | 0.74 | | |
| **Fe** | | | | | | | | | | | 0.46 | | | 0.67 | | 0.71 | 0.52 | | 0.40 |
| **K** | | | | | | | | | | | | 0.61 | 0.46 | 0.72 | | 0.55 | 0.54 | | |
| **Li** | | | | | | | | | | | | | | 0.31 | | 0.44 | 0.49 | | |
| **Mg** | | | | | | | | | | | | | | 0.52 | 0.50 | 0.38 | 0.34 | | 0.51 |
| **Mn** | | | | | | | | | | | | | | | | 0.58 | 0.40 | | 0.30 |
| **Na** | | | | | | | | | | | | | | | | | | -0.36 | 0.51 |
| **Ni** | | | | | | | | | | | | | | | | | 0.74 | 0.38 | 0.35 |
| **Pb** | | | | | | | | | | | | | | | | | | | |
| **Sr** | | | | | | | | | | | | | | | | | | | |
| **Zn** | | | | | | | | | | | | | | | | | | | |

### 3.8.4 Inorganic contaminant concentrations in textiles

**Local variability of inorganic contaminant concentrations in the textile samples**

Metal concentrations in the Hadleigh Marsh landfill textile samples determined by ICP-OES analysis are shown in Figure 3.15 split by trial pit. The metal concentrations in the textile samples from the two trial pits were highly heterogeneous. Metal concentrations in trial pit 3 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to an order of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to five orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 3. Metal concentrations in trial pit 4 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to two orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals in trial pit 4.

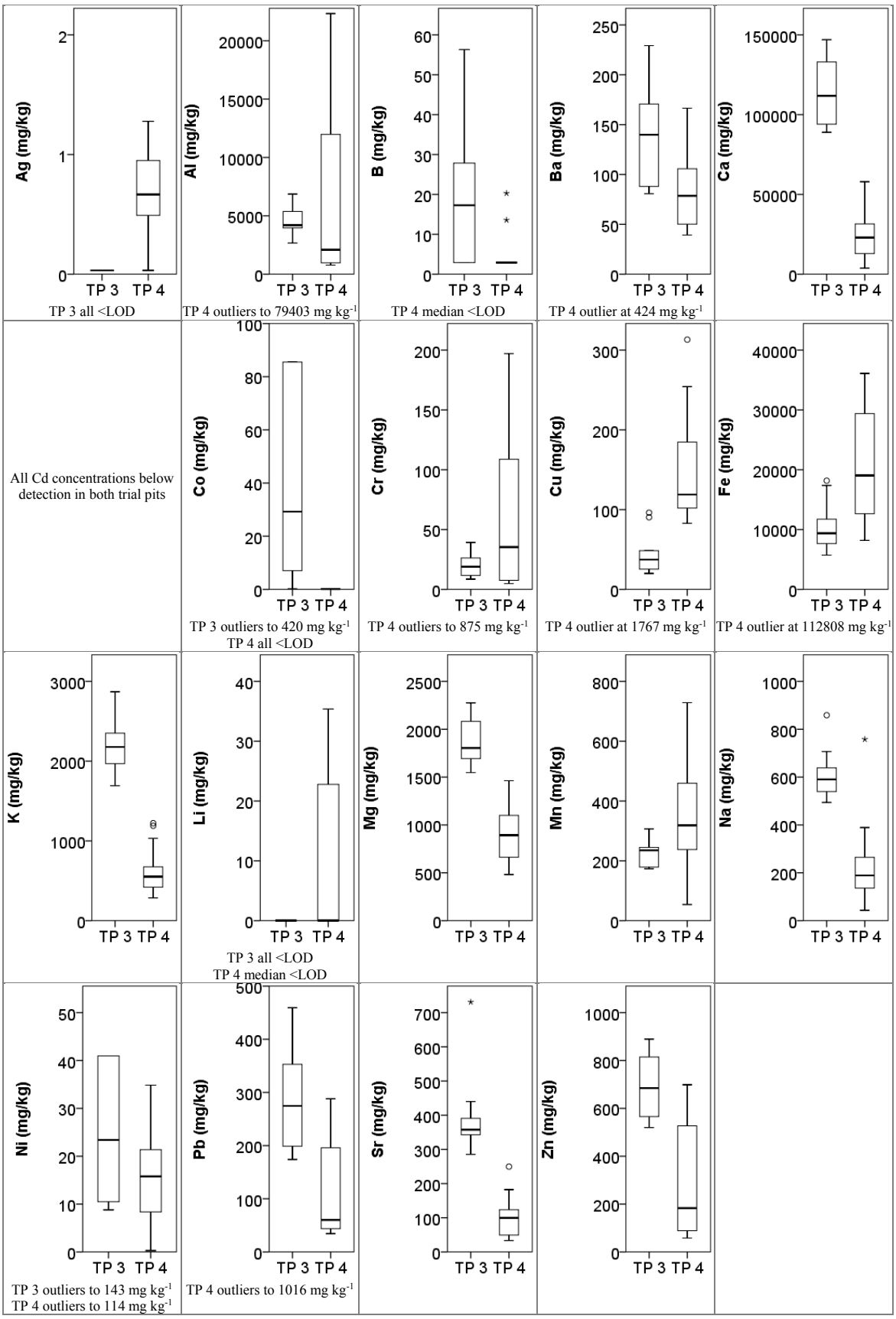



**Figure 3.15: Inorganic contaminant concentrations in the textile samples from the Hadleigh Marsh trial pits (trial pit 3, n=9; trial pit 4, n=18)**

### Variability of inorganic contaminant concentrations in Hadleigh Marsh textiles

To test whether there were differences in contaminant concentrations in the textile samples between the two trial pits within Hadleigh Marsh, statistical comparisons (Mann-Whitney U tests) were carried out. The following metals were not included in the statistical analysis as more than 50% of the data were below the LOD: Cd and Li. Table 3.23 summarises the results of these comparisons, Al, Cr and Ni were not significantly different between the two trial pits. Trial pit 3 had significantly higher concentrations of B, Ba, Ca, Co, K, Mg, Na, Pb, Sr and Zn in the textile samples compared to trial pit 4. Trial pit 4 had significantly higher concentrations of Ag, Cu, Fe and Mn in the textile samples compared to trial pit 3.

**Table 3.29: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in inorganic contaminant concentrations in textile samples between the two trial pit locations within Hadleigh Marsh landfill (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Trial pit 3 median (mg $kg^{-1}$) (n=9) | Trial pit 4 median (mg $kg^{-1}$) (n=18) |
|---|---|---|---|---|
| Ag | <0.001 | TP3<TP4 | <LOD | 0.67 |
| Al | 0.504 | No | 4202 | 2090 |
| B | 0.002 | TP3>TP4 | 17.2 | <LOD |
| Ba | 0.012 | TP3>TP4 | 140 | 79 |
| Ca | <0.001 | TP3>TP4 | 111833 | 22932 |
| Cd | insufficient data | | <LOD | <LOD |
| Co | <0.001 | TP3>TP4 | 29.2 | <LOD |
| Cr | 0.607 | No | 19.0 | 35.3 |
| Cu | <0.001 | TP3<TP4 | 37.4 | 119 |
| Fe | 0.003 | TP3<TP4 | 9384 | 19053 |
| K | <0.001 | TP3>TP4 | 2177 | 552 |
| Li | insufficient data | | <LOD | <LOD |
| Mg | <0.001 | TP3>TP4 | 1804 | 893 |
| Mn | 0.045 | TP3<TP4 | 235 | 319 |
| Na | <0.001 | TP3>TP4 | 590 | 189 |
| Ni | 0.123 | No | 23.415 | 15.772 |
| Pb | 0.006 | TP3>TP4 | 275 | 60.2 |
| Sr | <0.001 | TP3>TP4 | 358 | 100 |
| Zn | 0.001 | TP3>TP4 | 685 | 183 |

Table 3.30 shows the metal concentrations in the textile waste samples for the overall site, i.e. data for trial pits 3 and 4 combined. Metal concentrations in the textiles were highly heterogeneous, differing by up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, and up to two orders of magnitude between the minimum and maximum concentrations of each of the major metals. There were also differences of up to four orders of magnitude between the median concentrations of different trace metals and differences of up to two orders of magnitude between the median concentrations of different major metals.

**Table 3.30: Inorganic contaminant concentrations in <10 mm fraction of textile waste samples from Hadleigh Marsh (n=27)**

| Element | Limit of Detection (mg $kg^{-1}$) | Minimum (mg $kg^{-1}$) | Maximum (mg $kg^{-1}$) | Range (mg $kg^{-1}$) | Median (mg $kg^{-1}$) |
|---|---|---|---|---|---|
| Ag | 0.06 | <LOD | 1.28 | 1.3 | 0.49 |
| Al | 0.12 | 778 | 79403 | 78625 | 4051 |
| B | 5.83 | <LOD | 56 | 53 | <LOD |
| Ba | 0.09 | 39 | 424 | 385 | 87 |
| Ca | 0.16 | 3762 | 147027 | 143265 | 31507 |
| Cd | 0.72 | <LOD | <LOD | - | <LOD |
| Co | 0.31 | <LOD | 421 | 421 | <LOD |
| Cr | 0.12 | 4.80 | 875 | 870 | 19 |
| Cu | 0.08 | 20 | 1767 | 1747 | 102 |
| Fe | 6.55 | 5719 | 112808 | 107089 | 13847 |
| K | 0.17 | 286 | 2870 | 2584 | 676 |
| Li | 0.03 | <LOD | 35 | 35 | <LOD |
| Mg | 0.09 | 482 | 2276 | 1794 | 1100 |
| Mn | 0.07 | 54 | 729 | 675 | 273 |
| Na | 0.07 | 43 | 859 | 816 | 264 |
| Ni | 0.60 | <LOD | 143 | 143 | 17 |
| Pb | 1.11 | 34 | 1016 | 982 | 174 |
| Sr | 0.05 | 33 | 731 | 698 | 123 |
| Zn | 0.17 | 58 | 889 | 831 | 522 |

## Correlation analyses

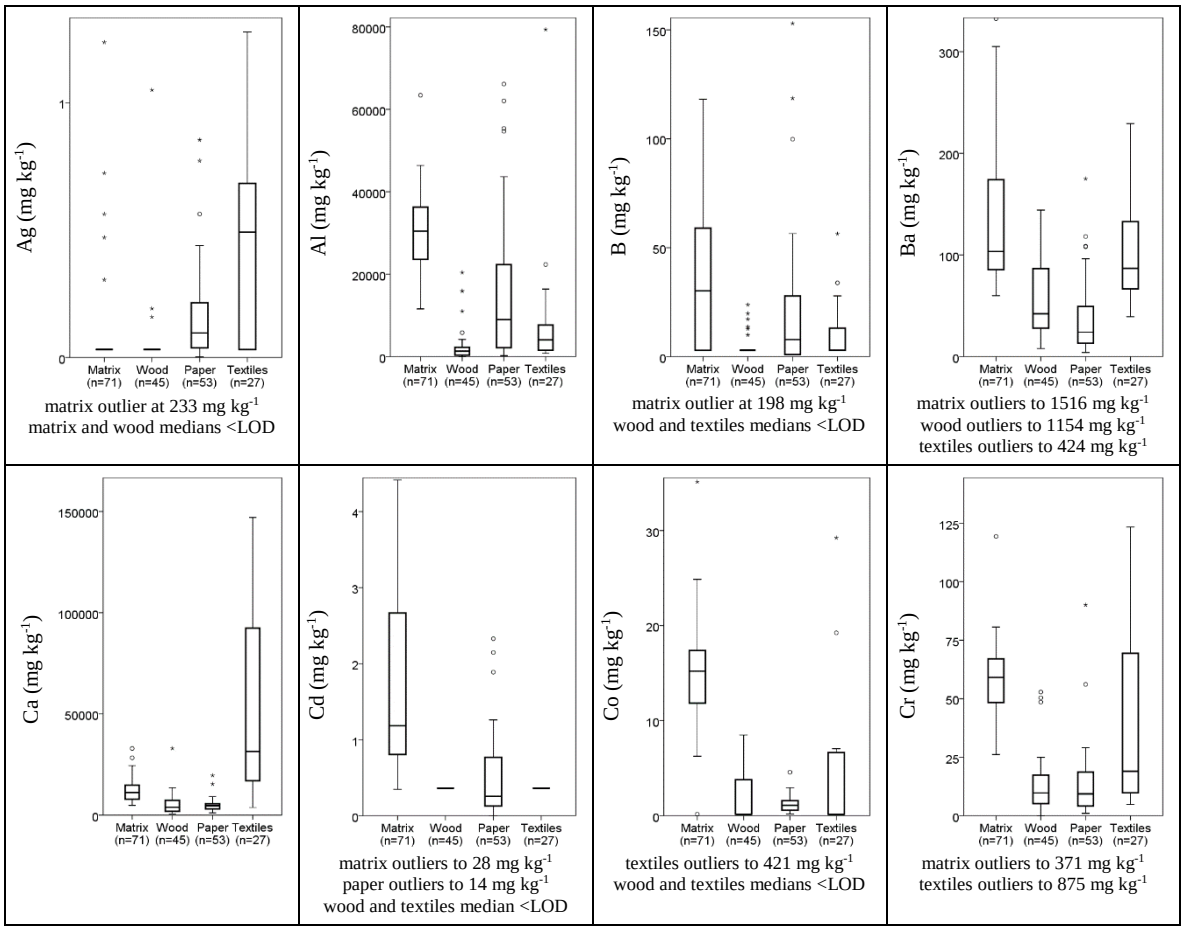
Spearman's rank correlation coefficients are shown in Table 3.31 for tests to determine whether there were correlations between metal concentrations or between metal concentrations and moisture content in the Hadleigh Marsh textile samples. Strong correlations between metals could indicate they are from a common source, and correlations between metals and moisture content could indicate that the metal concentrations have been influenced by leachate movement. Chromium showed a moderate positive correlation with moisture content, and Ag, Cu, and Fe showed strong positive correlations with moisture content. Calcium and Sr showed strong negative correlations with moisture content. There were many correlations between the metals, both positive and negative, but few strong correlations were found.

**Table 3.31: Spearman's rank correlation coefficients for metals and moisture content (MC) in Hadleigh Marsh textile samples. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8. B, Cd, Co and Li omitted as median values<LOD**

| | Ag | Al | Ba | Ca | Cr | Cu | Fe | K | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **MC** | **0.90** | | | **-0.85** | 0.68 | **0.92** | **0.83** | | | | | | | **-0.80** | |
| **Ag** | | | | -0.71 | 0.39 | 0.61 | 0.59 | -0.52 | -0.44 | | -0.41 | | | -0.58 | |
| **Al** | | | 0.48 | | 0.41 | | | 0.55 | | | 0.44 | | | | 0.39 |
| **Ba** | | | | | 0.56 | | | 0.60 | 0.58 | -0.46 | 0.58 | 0.68 | 0.73 | | 0.71 |
| **Ca** | | | | | -0.45 | **-0.81** | -0.54 | 0.67 | 0.76 | | 0.68 | | | **0.92** | |
| **Cr** | | | | | | | 0.62 | | | | | **0.80** | 0.45 | | 0.60 |
| **Cu** | | | | | | | 0.51 | -0.70 | -0.76 | | -0.64 | | | **-0.88** | |
| **Fe** | | | | | | | | | | 0.46 | | | | -0.43 | |
| **K** | | | | | | | | | **0.88** | | **0.88** | | | 0.74 | 0.72 |
| **Mg** | | | | | | | | | | | **0.95** | 0.39 | | **0.85** | 0.64 |
| **Mn** | | | | | | | | | | | | | -0.60 | | |
| **Na** | | | | | | | | | | | | 0.43 | | 0.76 | 0.72 |
| **Ni** | | | | | | | | | | | | | 0.67 | | **0.80** |
| **Pb** | | | | | | | | | | | | | | | 0.65 |
| **Sr** | | | | | | | | | | | | | | | 0.45 |
| **Zn** | | | | | | | | | | | | | | | |

### 3.8.5 Comparison of the inorganic contaminant concentrations in the matrix material, wood, paper and textiles

Figure 3.16 compares the contaminant concentrations in the different materials analysed. Table 3.32 summarises the statistical analyses that were carried out to determine whether there were significant differences in contaminant concentrations between the matrix material, wood, paper and textiles, and shows the ranking of the contaminant concentrations in the materials for each element (Kruskal-Wallis test, all $p<0.001$, and post-hoc analyses using Mann-Whitney U tests). Paper typically contained the lowest concentrations of metals and the matrix material the highest; however, there were some exceptions: Zn concentrations were approximately 500% higher (median values) in the wood samples compared to the matrix material. Ca and Cu were present in concentrations approximately 300% higher (median values) in textiles compared to the matrix material, and Sr concentrations were approximately 50% higher in textiles than in the matrix material.



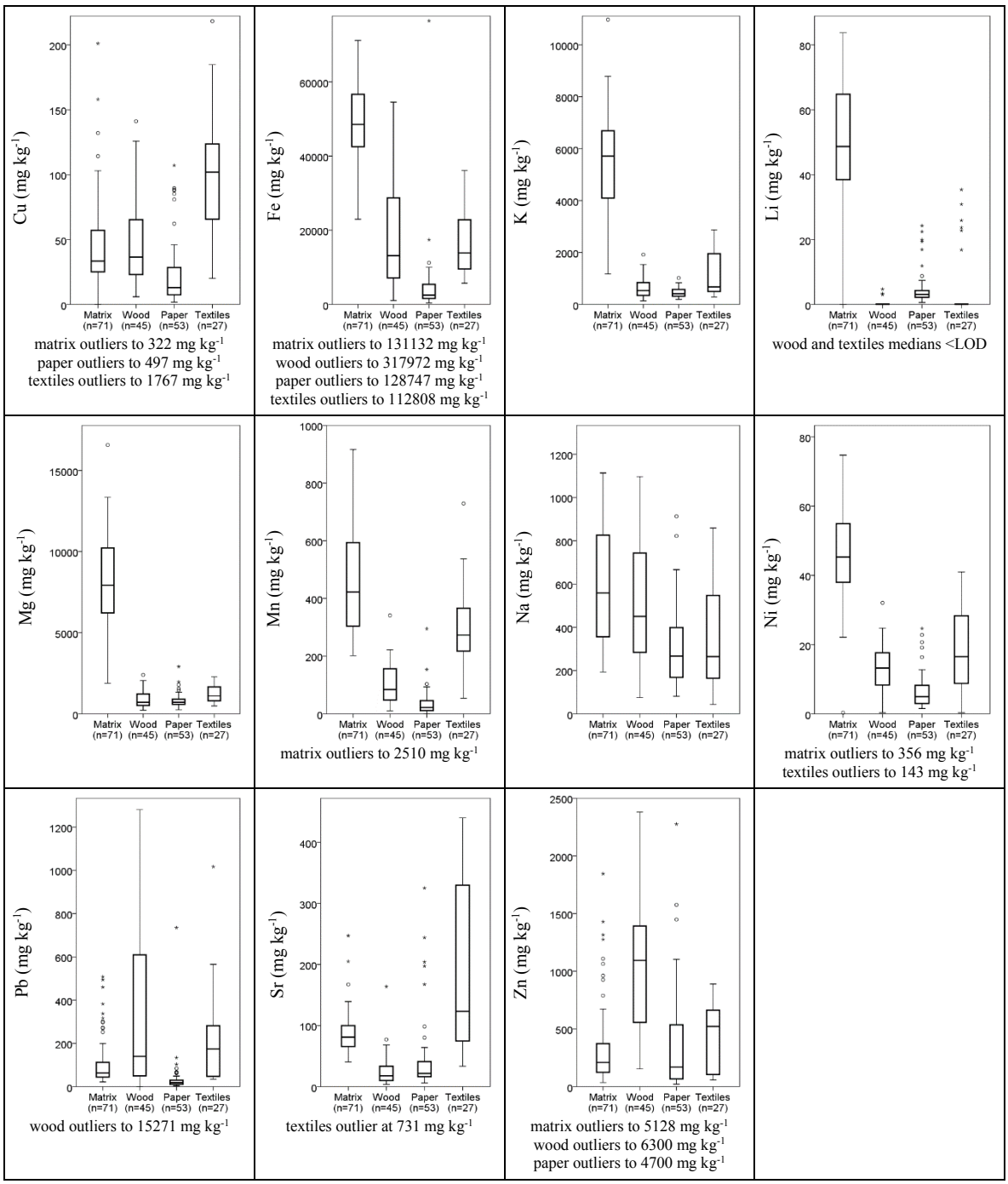



**Figure 3.16: Boxplots showing inorganic contaminant concentrations in the matrix material, wood, paper and textile samples from Hadleigh Marsh landfill**

**Table 3.32: Summary of the post-hoc Mann-Whitney U tests to determine the relative contaminant concentration in the different materials. Median values and sample numbers can be found in Table 3.20, Table 3.24, Table 3.27 and Table 3.30 (critical p value=0.0083 with Bonferroni correction)**

| Element | Matrix vs Wood | Matrix vs Paper | Matrix vs Textiles | Wood vs Paper | Wood vs Textiles | Paper vs Textiles | Summary |
|---|---|---|---|---|---|---|---|
| Ag | medians <LOD | medians <LOD | medians <LOD | medians <LOD | medians <LOD | P=T (p=0.010) | insufficient data to determine ranking |
| Al | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | P>W (p<0.001) | T>W (p<0.001) | P=T (p=0.050) | M>P=T>W |
| B | medians <LOD | M>P (p<0.001) | M>T (p<0.001) | medians <LOD | medians <LOD | medians <LOD | insufficient data to determine ranking |
| Ba | M>W (p<0.001) | M>P (p<0.001) | M=T (p=0.040) | W>P (p=0.004) | T>W (p=0.001) | T>P (p<0.001) | M=T>W>P |
| Ca | M>W (p<0.001) | M>P (p<0.001) | T>M (p<0.001) | W=P (p=0.867) | T>W (p<0.001) | T>P (p<0.001) | T>M>P=W |
| Cd | medians <LOD | M>P (p<0.001) | medians <LOD | medians <LOD | medians <LOD | medians <LOD | insufficient data to determine ranking |
| Co | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | medians <LOD | medians <LOD | medians <LOD | insufficient data to determine ranking |
| Cr | M>W (p<0.001) | M>P (p<0.001) | M>T (p=0.002) | W=P (p=0.957) | T>W (p=0.001) | T>P (p=0.001) | M>T>W=P |
| Cu | M=W (p=0.890) | M>P (p<0.001) | T>M (p<0.001) | W>P (p<0.001) | T>W (p<0.001) | T>P (p<0.001) | T>W=M>P |
| Fe | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | W>P (p<0.001) | W=T (p=0.767) | T>P (p<0.001) | M>T=W>P |
| K | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | W=P (p=0.010) | W=T (p=0.018) | T>P (p<0.001) | insufficient data to determine ranking |
| Li | medians <LOD | M>P (p<0.001) | medians <LOD | medians <LOD | medians <LOD | medians <LOD | insufficient data to determine ranking |
| Mg | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | W=P (p=0.980) | T=W (p=0.010) | T>P (p<0.001) | insufficient data to determine ranking |
| Mn | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | W>P (p<0.001) | T>W (p<0.001) | T>P (p<0.001) | M>T>W>P |
| Na | M=W (p=0.067) | M>P (p<0.001) | M>T (p<0.001) | W>P (p=0.001) | W=T (p=0.046) | P=T (p=0.499) | insufficient data to determine ranking |
| Ni | M>W (p<0.001) | M>P (p<0.001) | M>T (p<0.001) | W>P (p<0.001) | W=T (p=0.070) | T>P (p<0.001) | M>T=W>P |
| Pb | M=W (p=0.013) | M>P (p<0.001) | M=T (p=0.050) | W>P (p<0.001) | T=W (p=0.663) | T>P (p<0.001) | T=W=M>P |
| Sr | M>W (p<0.001) | M>P (p<0.001) | T>M (p=0.004) | W=P (p=0.040) | T>W (p<0.001) | T>P (p<0.001) | T>M>P=W |
| Zn | W>M (p<0.001) | M=P (p=0.362) | M=T (p=0.225) | W>P (p<0.001) | W>T (p<0.001) | P=T (p=0.153) | W>T=M=P |

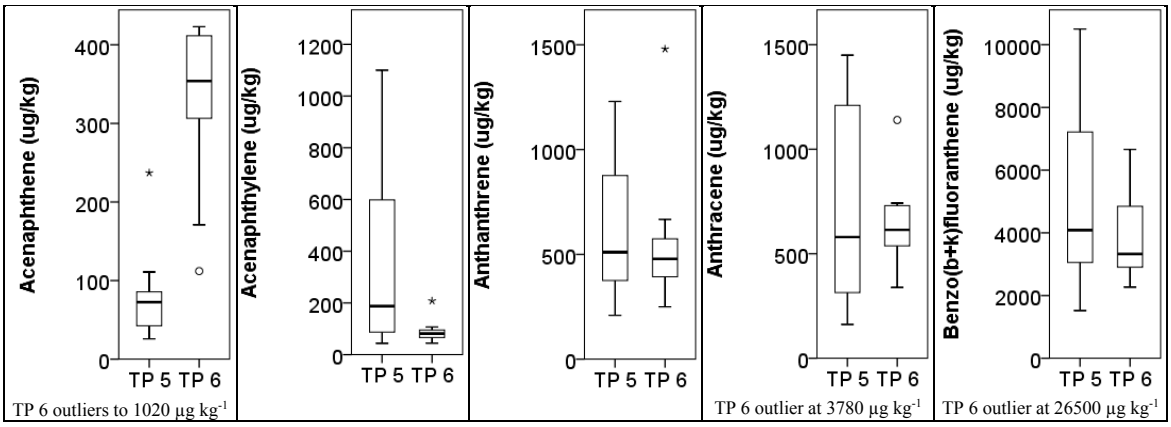
## 3.9 PAH concentrations in the waste samples

### 3.9.1 Organic contaminant concentrations in the matrix material

#### *Leigh Marshes*

**Local variability of organic contaminant concentrations in the matrix material**

PAH concentrations in the Leigh Marshes landfill matrix material samples determined by the EA National Laboratory using GC-MS analysis are shown in Figure 3.17 split by trial pit. One sample (6E-Matrix A) was the only outlier above the median for anthanthrene, benzo(b+k)fluoranthene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(e)pyrene, benzo(ghi)perylene, benzo(k)fluoranthene, coronene, dibenzo(ah)anthracene, fluoranthene, naphthalene, perylene and pyrene, and one of two outliers about the median for acenaphthene, anthracene, chrysene, fluorene, indeno(1,2,3-cd)pyrene and phenanthrene. The PAH concentrations in the matrix material samples from the two Leigh Marshes trial pits were highly heterogeneous. PAH concentrations in trial pit 5 vary by up to three orders of magnitude between the minimum and maximum concentrations of each of the PAHs. There were also differences of up to three orders of magnitude between the median concentrations of different PAHs in trial pit 5. PAH concentrations in trial pit 6 vary by up to two orders of magnitude between the minimum and maximum concentrations of each of the PAHs. There were also differences of up to three orders of magnitude between the median concentrations of different PAHs in trial pit 6.



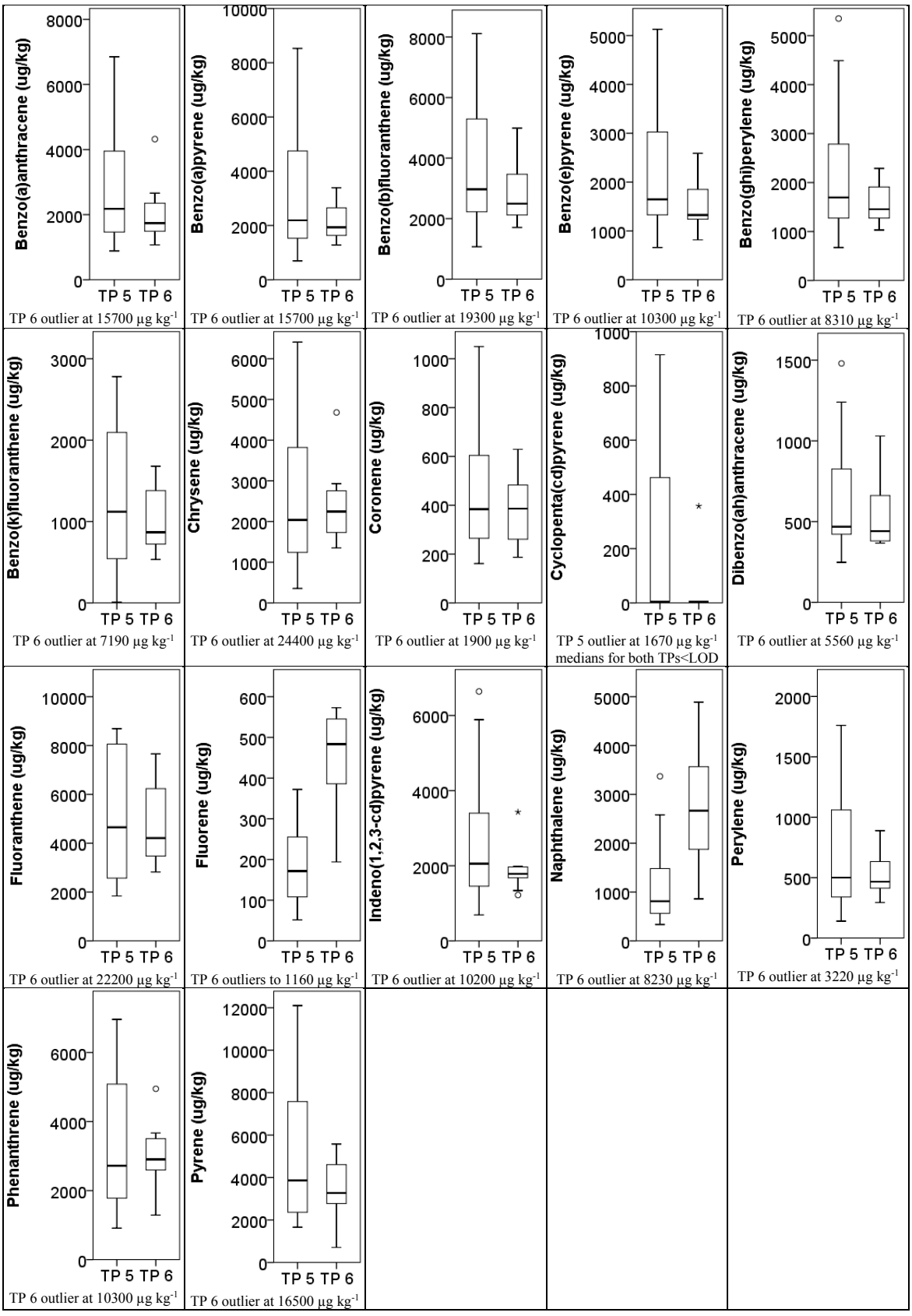



**Figure 3.17: Organic contaminant concentrations in the matrix material from the Leigh Marshes trial pits (trial pit 5, n=12; trial pit 6, n=12)**

**Variability of the matrix material's organic contaminant concentrations within Leigh Marshes**

To test whether there were also differences in contaminant concentrations within individual sites, statistical comparisons (Mann-Whitney U tests) between the two trial pits in each site were carried out. Cyclopenta(cd)pyrene was not included in the statistical analyses as more than 50% of the samples were below the LOD. Table 3.33 summarises the results of these comparisons for Leigh Marshes. Only acenaphthene, acenaphthylene, fluorene and naphthalene concentrations were significantly different between the two trial pits within Leigh Marshes landfill, acenaphthylene was highest in trial pit 5, the others were highest in trial pit 6.

**Table 3.33: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in organic contaminant concentrations in the matrix material between the two trial pit locations within Leigh Marshes landfill (critical p value = 0.05)**

| | Mann-Whitney U p value | Significant difference | Trial pit 5 median (n=12) (µg $kg^{-1}$) | Trial pit 6 median (n=12) (µg $kg^{-1}$) |
|---|---|---|---|---|
| acenaphthene | <0.001 | TP5<TP6 | 73 | 354 |
| acenaphthylene | 0.039 | TP5>TP6 | 188 | 82 |
| anthanthrene | 0.755 | no | 511 | 479 |
| anthracene | 0.551 | no | 580 | 614 |
| benzo(b+k)fluoranthene | 0.478 | no | 4085 | 3325 |
| benzo(a)anthracene | 0.410 | no | 2180 | 1740 |
| benzo(a)pyrene | 0.755 | no | 2190 | 1935 |
| benzo(b)fluoranthene | 0.630 | no | 2970 | 2495 |
| benzo(e)pyrene | 0.291 | no | 1645 | 1325 |
| benzo(ghi)perylene | 0.630 | no | 1695 | 1455 |
| benzo(k)fluoranthene | 0.799 | no | 1120 | 868 |
| chrysene | 0.671 | no | 2040 | 2245 |
| coronene | 0.932 | no | 384 | 386 |
| cyclopenta(cd)pyrene | Insufficient data | | <10 | <10 |
| dibenzo(ah)anthracene | 0.551 | no | 468 | 441 |
| fluoranthene | 0.932 | no | 4650 | 4210 |
| fluorene | <0.001 | TP5<TP6 | 172 | 484 |
| indeno(1,2,3-cd)pyrene | 0.630 | no | 2055 | 1785 |
| naphthalene | 0.001 | TP5<TP6 | 812 | 2665 |
| perylene | 0.932 | no | 501 | 466 |
| phenanthrene | 0.551 | no | 2720 | 2905 |
| pyrene | 0.713 | no | 3865 | 3275 |

To investigate further the heterogeneity of the waste within Leigh Marshes landfill, the contaminants that were assessed in the matrix material both in this research and Halcrow's previous study in 2012 are compared in Table 3.34. Statistical analysis (Mann-Whitney U tests) of the datasets for Leigh Marshes showed there to be a significant difference in measured concentrations of some contaminants between the two studies, with this research finding higher concentrations of benzo(b)fluoranthene ($p=0.004$) fluoranthene ($p=0.032$), fluorene ($p=0.031$), naphthalene ($p<0.001$), phenanthrene ($p=0.005$) and pyrene (0.042).

**Table 3.34: Organic contaminant concentrations found in the Leigh Marshes matrix material in this research compared to the Halcrow study**

| | This research (n=24) | | | Halcrow study (n=30) | | |
|---|---|---|---|---|---|---|
| | Min (µg kg$^{-1}$) | Max (µg kg$^{-1}$) | Median (µg kg$^{-1}$) | Min (µg kg$^{-1}$) | Max (µg kg$^{-1}$) | Median (µg kg$^{-1}$) |
| acenaphthene | 25.8 | 1020 | 142 | <100 | 3897 | <100 |
| acenaphthylene | 44 | 1100 | 95 | <100 | 2454 | 113 |
| anthracene | 162 | 3780 | 606 | <100 | 12114 | 240 |
| benzo(a)anthracene | 887 | 15700 | 2010 | <100 | 29157 | 1281 |
| benzo(a)pyrene | 696 | 15700 | 2080 | <100 | 28240 | 1250 |
| benzo(b)fluoranthene | 1070 | 19300 | 2910 | <100 | 24723 | 1279 |
| benzo(ghi)perylene | 672 | 8310 | 1570 | <100 | 18032 | 1283 |
| benzo(k)fluoranthene | <20 | 7190 | 930 | <100 | 21935 | 1065 |
| chrysene | 353 | 24400 | 2040 | <100 | 30234 | 924 |
| dibenzo(ah)anthracene | 248 | 5560 | 451 | <100 | 4688 | 304 |
| fluoranthene | 1840 | 22200 | 4405 | <100 | 65074 | 1593 |
| fluorene | 51.4 | 1160 | 291 | <100 | 4294 | 168 |
| indeno(1,2,3-cd)pyrene | 690 | 10200 | 1885 | <100 | 19978 | 1340 |
| naphthalene | 335 | 8230 | 1840 | <100 | 1207 | <100 |
| phenanthrene | 913 | 10300 | 2770 | <100 | 43077 | 459 |
| pyrene | 709 | 16500 | 3475 | <100 | 52490 | 1752 |

Table 3.35 shows the PAH concentrations in the matrix material for the overall site, i.e. data for trial pits 5 and 6 combined. Organic contaminant concentrations in the matrix material were highly heterogeneous differing by up to three orders of magnitude between the minimum and maximum concentrations measured in samples, e.g. chrysene. There were also differences of up to three orders of magnitude between the median concentrations of different organic contaminants, e.g. cyclopenta(cd)pyrene compared to fluoranthene.

**Table 3.35: Organic contaminant concentrations in <2 mm fraction of matrix material waste samples from Leigh Marshes landfill**

| | Leigh Marshes (n=24) | | | |
|---|---|---|---|---|
| | **Min. (µg kg$^{-1}$)** | **Max. (µg kg$^{-1}$)** | **Range (µg kg$^{-1}$)** | **Median (µg kg$^{-1}$)** |
| acenaphthene | 25.8 | 1020 | 994 | 142 |
| acenaphthylene | 44 | 1100 | 1056 | 95 |
| anthanthrene | 209 | 1480 | 1271 | 479 |
| anthracene | 162 | 3780 | 3618 | 606 |
| benzo(b+k)fluoranthene | 1520 | 26500 | 24980 | 3755 |
| benzo(a)anthracene | 887 | 15700 | 14813 | 2010 |
| benzo(a)pyrene | 696 | 15700 | 15004 | 2080 |
| benzo(b)fluoranthene | 1070 | 19300 | 18230 | 2910 |
| benzo(e)pyrene | 660 | 10300 | 9640 | 1575 |
| benzo(ghi)perylene | 672 | 8310 | 7638 | 1570 |
| benzo(k)fluoranthene | <20 | 7190 | 7180 | 930 |
| chrysene | 353 | 24400 | 24047 | 2040 |
| coronene | 161 | 1900 | 1739 | 386 |
| cyclopenta(cd)pyrene | <10 | 1670 | 1665 | <10 |
| dibenzo(ah)anthracene | 248 | 5560 | 5312 | 451 |
| fluoranthene | 1840 | 22200 | 20360 | 4405 |
| fluorene | 51.4 | 1160 | 1109 | 291 |
| indeno(1,2,3-cd)pyrene | 690 | 10200 | 9510 | 1885 |
| naphthalene | 335 | 8230 | 7895 | 1840 |
| perylene | 140 | 3220 | 3080 | 466 |
| phenanthrene | 913 | 10300 | 9387 | 2770 |
| pyrene | 709 | 16500 | 15791 | 3475 |

## Correlation analyses

Strong correlations between PAHs could indicate they are from a common source, and correlations between PAHs and moisture content could indicate that the PAH concentrations have been influenced by leachate movement. No correlations were found between PAHs and moisture content in Leigh Marshes matrix material. Spearman's rank correlation coefficients are shown in Table 3.36 for tests to determine whether there were correlations between different PAHs in Leigh Marshes matrix material. There were moderate or strong correlations between most of the PAHs, exceptions were acenaphthene, fluorene and naphthalene which had few correlations with other PAHs.

**Table 3.36: Spearman's rank correlation coefficients for PAHs in Leigh Marshes matrix material. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8. Cyclopenta(cd)pyrene omitted as median value<LOD**

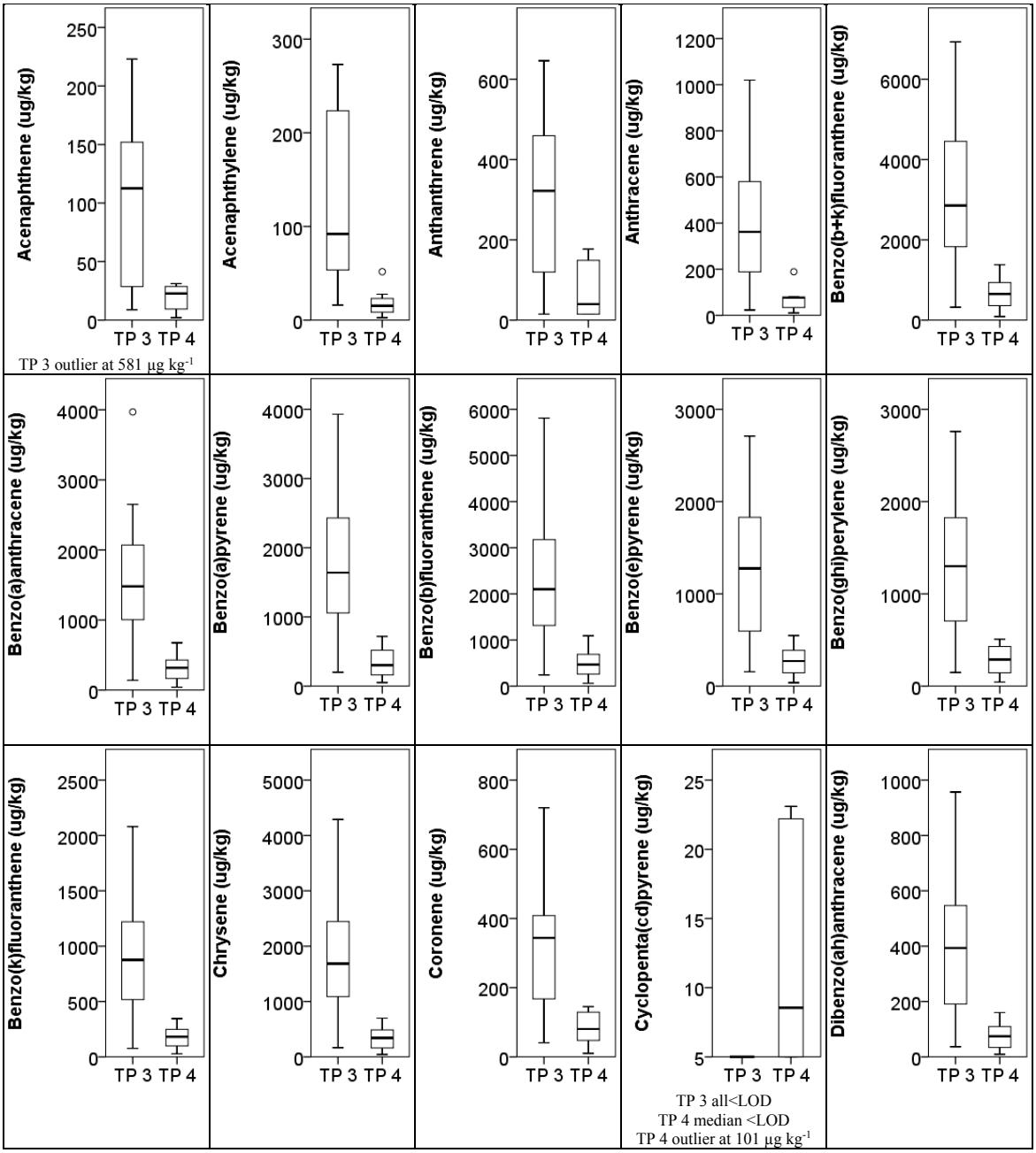
| | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **acenaphthene** | | | | 0.44 | | | | | | | | | | | | **0.94** | | 0.76 | | 0.46 | |
| **acenaphthylene** | | | 0.41 | | 0.44 | 0.42 | | 0.42 | 0.44 | | | 0.55 | | 0.51 | | | | | 0.48 | | 0.44 |
| **anthanthrene** | | | | 0.78 | **0.94** | **0.91** | **0.94** | **0.92** | **0.87** | **0.94** | **0.94** | 0.77 | **0.82** | 0.73 | **0.90** | | **0.90** | | **0.87** | **0.80** | **0.82** |
| **anthracene** | | | | | 0.74 | 0.79 | 0.77 | 0.75 | 0.71 | 0.75 | 0.78 | 0.71 | 0.71 | 0.69 | **0.89** | 0.65 | 0.71 | 0.49 | 0.72 | **0.94** | 0.75 |
| **benzo(b+k)fluoranthene** | | | | | | **0.97** | **0.98** | **0.99** | **0.94** | **0.98** | **0.95** | **0.82** | **0.91** | 0.75 | **0.90** | | **0.88** | | **0.86** | 0.73 | **0.81** |
| **benzo(a)anthracene** | | | | | | | **0.96** | **0.96** | **0.93** | **0.96** | **0.94** | **0.81** | **0.86** | 0.70 | **0.92** | | **0.89** | | **0.88** | 0.75 | **0.83** |
| **benzo(a)pyrene** | | | | | | | | **0.97** | **0.89** | **0.99** | **0.95** | 0.80 | **0.92** | 0.73 | **0.91** | | **0.88** | | **0.85** | 0.73 | **0.82** |
| **benzo(b)fluoranthene** | | | | | | | | | **0.93** | **0.97** | **0.94** | **0.82** | **0.93** | 0.73 | **0.89** | | **0.84** | | **0.84** | 0.73 | 0.80 |
| **benzo(e)pyrene** | | | | | | | | | | **0.89** | **0.86** | 0.75 | **0.82** | 0.72 | **0.83** | | **0.84** | | **0.83** | 0.71 | 0.76 |
| **benzo(ghi)perylene** | | | | | | | | | | | **0.96** | **0.82** | **0.93** | 0.74 | **0.90** | | **0.91** | | **0.86** | 0.72 | **0.81** |
| **benzo(k)fluoranthene** | | | | | | | | | | | | **0.82** | **0.87** | 0.68 | **0.92** | | **0.93** | | **0.92** | 0.77 | **0.82** |
| **chrysene** | | | | | | | | | | | | | 0.80 | 0.64 | 0.74 | | 0.71 | 0.45 | **0.88** | 0.65 | 0.68 |
| **coronene** | | | | | | | | | | | | | | 0.78 | 0.79 | | 0.77 | | 0.75 | 0.67 | 0.69 |
| **dibenzo(ah)anthracene** | | | | | | | | | | | | | | | 0.64 | | 0.70 | | 0.60 | 0.66 | 0.74 |
| **fluoranthene** | | | | | | | | | | | | | | | | 0.48 | **0.84** | | **0.81** | **0.89** | **0.84** |
| **fluorene** | | | | | | | | | | | | | | | | | | 0.73 | | 0.64 | |
| **indeno(1,2,3-cd)pyrene** | | | | | | | | | | | | | | | | | | | **0.89** | 0.69 | 0.77 |
| **naphthalene** | | | | | | | | | | | | | | | | | | | | 0.45 | |
| **perylene** | | | | | | | | | | | | | | | | | | | | 0.69 | 0.75 |
| **phenanthrene** | | | | | | | | | | | | | | | | | | | | | 0.72 |
| **pyrene** | | | | | | | | | | | | | | | | | | | | | |

## *Hadleigh Marsh*

### Local variability of organic contaminant concentrations in the matrix material

PAH concentrations in the Hadleigh Marsh landfill matrix material samples determined by the EA National Laboratory using GC-MS analysis are shown in Figure 3.18 split by trial pit. The PAH concentrations in the matrix material samples from the two Hadleigh Marsh trial pits were highly heterogeneous. PAH concentrations in trial pit 3 vary by up to two orders of magnitude between the minimum and maximum concentrations of each of the PAHs. There were also differences of up to three orders of magnitude between

the median concentrations of different PAHs in trial pit 3. PAH concentrations in trial pit 4 vary by up to two orders of magnitude between the minimum and maximum concentrations of each of the PAHs. There were also differences of up to two orders of magnitude between the median concentrations of different PAHs in trial pit 4.



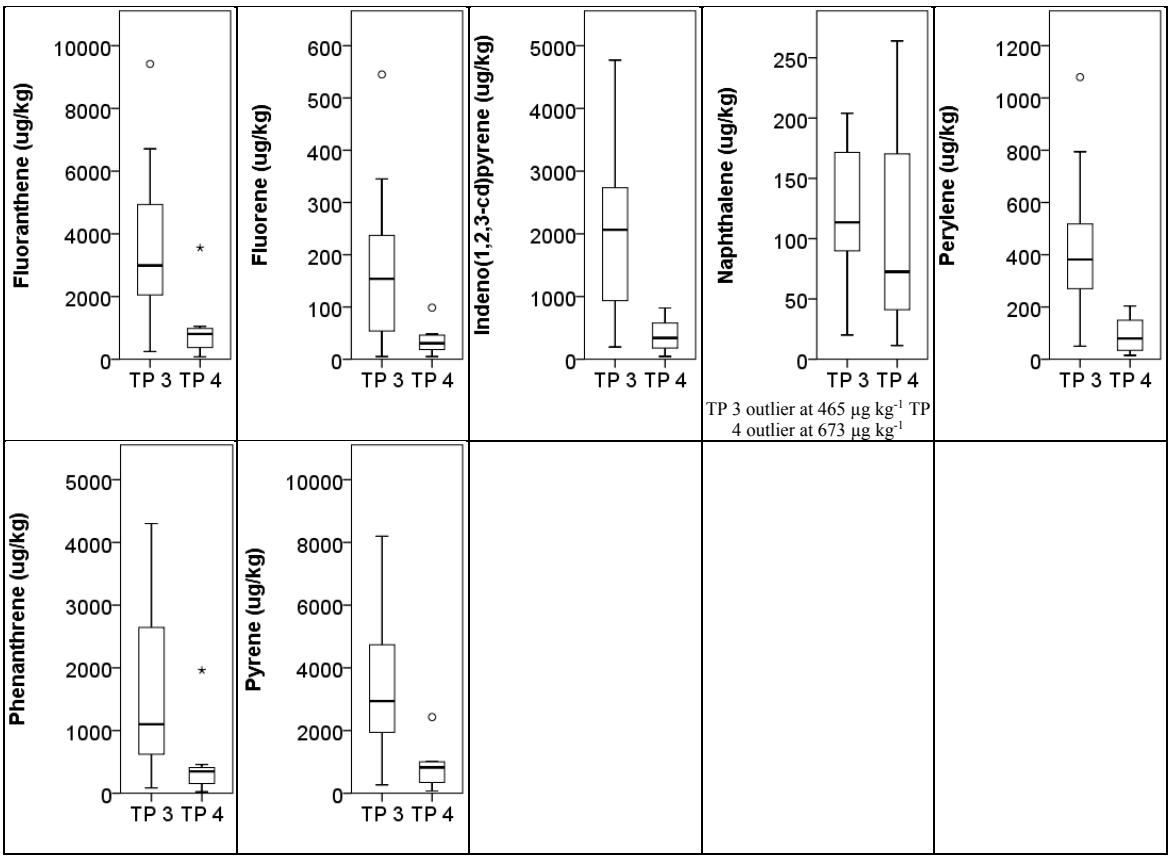



**Figure 3.18: Organic contaminant concentrations in the matrix material from the Hadleigh Marsh trial pits (trial pit 3, n=12; trial pit 4, n=8)**

## Variability of the matrix material's organic contaminant concentrations within Hadleigh Marsh

To test whether there were also differences in contaminant concentrations within individual sites, statistical comparisons (Mann-Whitney U tests) between the two trial pits in each site were carried out. Cyclopenta(cd)pyrene was not included in the statistical analyses as more than 50% of the data were below the LOD. Table 3.37 summarises the results of these comparisons for Hadleigh Marsh. In Hadleigh Marshes all PAH concentrations, with the exception of naphthalene, were consistently significantly higher in trial pit 3.

**Table 3.37: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in organic contaminant concentrations in the matrix material between the two trial pit locations within Hadleigh Marsh landfill (critical p value = 0.05)**

| Contaminant | Mann-Whitney U p value | Significant difference | Trial pit 3 median (n=12) (µg $kg^{-1}$) | Trial pit 4 median (n=8) (µg $kg^{-1}$) |
|---|---|---|---|---|
| acenaphthene | 0.005 | TP3>TP4 | 113 | 23 |
| acenaphthylene | <0.001 | TP3>TP4 | 92 | 15 |
| anthanthrene | 0.020 | TP3>TP4 | 322 | 40 |
| anthracene | 0.002 | TP3>TP4 | 362 | 76 |
| benzo(b+k)fluoranthene | 0.002 | TP3>TP4 | 2855 | 650 |
| benzo(a)anthracene | 0.002 | TP3>TP4 | 1480 | 318 |
| benzo(a)pyrene | 0.001 | TP3>TP4 | 1640 | 303 |
| benzo(b)fluoranthene | 0.003 | TP3>TP4 | 2100 | 467 |
| benzo(e)pyrene | 0.002 | TP3>TP4 | 1275 | 271 |
| benzo(ghi)perylene | 0.002 | TP3>TP4 | 1300 | 288 |
| benzo(k)fluoranthene | 0.003 | TP3>TP4 | 876 | 182 |
| chrysene | 0.001 | TP3>TP4 | 1685 | 342 |
| coronene | 0.004 | TP3>TP4 | 344 | 81 |
| cyclopenta(cd)pyrene | Insufficient data | | <10 | <10 |
| dibenzo(ah)anthracene | 0.003 | TP3>TP4 | 394 | 74 |
| fluoranthene | 0.016 | TP3>TP4 | 2990 | 806 |
| fluorene | 0.007 | TP3>TP4 | 154 | 31 |
| indeno(1,2,3-cd)pyrene | 0.001 | TP3>TP4 | 2065 | 340 |
| naphthalene | 0.208 | no | 114 | 73 |
| perylene | 0.001 | TP3>TP4 | 382 | 80 |
| phenanthrene | 0.007 | TP3>TP4 | 1100 | 348 |
| pyrene | 0.010 | TP3>TP4 | 2940 | 823 |

To investigate further the heterogeneity of the waste with Hadleigh Marsh landfill, the contaminants that have been assessed in the matrix material both in this research and Halcrow's previous study in 2012 are compared in Table 3.38. Statistical analysis (Mann-Whitney U tests) of the datasets for Hadleigh Marsh showed there to be a significant difference in measured concentrations of some contaminants between the two studies, with this research finding higher concentrations of benzo(b)fluoranthene (p=0.007) and indeno(123-cd)pyrene (p=0.021), and the Halcrow study finding higher concentrations of acenaphthene (p=0.001) and fluorene (p=0.022).

**Table 3.38: Organic contaminant concentrations found in the Hadleigh Marsh matrix material in this research compared to the Halcrow study**

| | This research (n=20) | | | Halcrow study (n=25) | | |
|---|---|---|---|---|---|---|
| | Min ($\mu g\ kg^{-1}$) | Max ($\mu g\ kg^{-1}$) | Median ($\mu g\ kg^{-1}$) | Min ($\mu g\ kg^{-1}$) | Max ($\mu g\ kg^{-1}$) | Median ($\mu g\ kg^{-1}$) |
| acenaphthene | <4 | 581 | 30 | <100 | 4311 | <100 |
| acenaphthylene | <5 | 273 | 44 | <100 | 967 | <100 |
| anthracene | <20 | 1020 | 186 | <100 | 3504 | <100 |
| benzo(a)anthracene | 40.5 | 3970 | 743 | <100 | 6655 | 181 |
| benzo(a)pyrene | 50.9 | 3930 | 761 | <100 | 4962 | 153 |
| benzo(b)fluoranthene | 60.5 | 5810 | 1055 | <100 | 4900 | 195 |
| benzo(ghi)perylene | 43.5 | 2760 | 577 | <100 | 3439 | 130 |
| benzo(k)fluoranthene | 27.4 | 2080 | 407 | <100 | 8296 | 130 |
| chrysene | 41.8 | 4290 | 771 | <100 | 6410 | 286 |
| dibenzo(ah)anthracene | 9.17 | 957 | 150 | <100 | 988 | <100 |
| fluoranthene | 74.7 | 9420 | 2050 | <100 | 15213 | 358 |
| fluorene | <10 | 545 | 54 | <100 | 3999 | 166 |
| indeno(1,2,3-cd)pyrene | 45.6 | 4770 | 854 | <100 | 4549 | 186 |
| naphthalene | 11.4 | 673 | 100 | <100 | 4572 | <100 |
| phenanthrene | 23.1 | 4300 | 622 | <100 | 11052 | 539 |
| pyrene | 72.1 | 8200 | 1940 | <100 | 11988 | 346 |

Table 3.39 shows the PAH concentrations in the matrix material for the overall site, i.e. data for trial pits 3 and 4 combined. Organic contaminant concentrations in the matrix material were highly heterogeneous differing by up to two orders of magnitude between the minimum and maximum concentrations measured in samples, e.g. phenanthrene. There were also differences of up to two orders of magnitude between the median concentrations of different organic contaminants, e.g. cyclopenta(cd)pyrene compared to fluoranthene.

**Table 3.39: Organic contaminant concentrations in <2 mm fraction of matrix material waste samples from Hadleigh Marsh landfill**

| | Hadleigh Marsh (n=20) | | | |
|---|---|---|---|---|
| | **Min. (µg kg$^{-1}$)** | **Max. (µg kg$^{-1}$)** | **Range (µg kg$^{-1}$)** | **Median (µg kg$^{-1}$)** |
| acenaphthene | <4 | 581 | 579 | 30 |
| acenaphthylene | <5 | 273 | 271 | 44 |
| anthanthrene | <30 | 646 | 631 | 156 |
| anthracene | <20 | 1020 | 1010 | 186 |
| benzo(b+k)fluoranthene | 87.9 | 6930 | 6842 | 1435 |
| benzo(a)anthracene | 40.5 | 3970 | 3930 | 743 |
| benzo(a)pyrene | 50.9 | 3930 | 3879 | 761 |
| benzo(b)fluoranthene | 60.5 | 5810 | 5750 | 1055 |
| benzo(e)pyrene | 36.8 | 2710 | 2673 | 567 |
| benzo(ghi)perylene | 43.5 | 2760 | 2717 | 577 |
| benzo(k)fluoranthene | 27.4 | 2080 | 2053 | 407 |
| chrysene | 41.8 | 4290 | 4248 | 771 |
| coronene | 10.3 | 720 | 710 | 144 |
| cyclopenta(cd)pyrene | <10 | 101 | 96 | <10 |
| dibenzo(ah)anthracene | 9.17 | 957 | 948 | 150 |
| fluoranthene | 74.7 | 9420 | 9345 | 2050 |
| fluorene | 5 | 545 | 540 | 54 |
| indeno(1,2,3-cd)pyrene | 45.6 | 4770 | 4724 | 854 |
| naphthalene | 11.4 | 673 | 662 | 100 |
| perylene | <30 | 1080 | 1065 | 212 |
| phenanthrene | 23.1 | 4300 | 4277 | 622 |
| pyrene | 72.1 | 8200 | 8128 | 1940 |

## Correlation analyses

Strong correlations between PAHs could indicate they are from a common source, and correlations between PAHs and moisture content could indicate that the PAH concentrations have been influenced by leachate movement. No correlations were found between PAHs and moisture content in Hadleigh Marsh matrix material. Spearman's rank correlation coefficients are shown in Table 3.40 for tests to determine whether there were correlations between different PAHs in Hadleigh Marsh matrix material. There were strong correlations between most of the PAHs, exceptions were

acenaphthene and naphthalene which predominantly had moderate correlations with other PAHs.

**Table 3.40: Spearman's rank correlation coefficients for PAHs in Hadleigh Marsh matrix material. Values shown are different from 0 with a significance level alpha=0.05. Underlined values show $r_s$>0.8. Cyclopenta(cd)pyrene omitted as median value<LOD**

| | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **acenaphthene** | | 0.77 | 0.72 | **0.91** | 0.79 | **0.83** | 0.79 | 0.77 | 0.77 | 0.75 | 0.74 | **0.81** | 0.80 | 0.77 | **0.85** | **0.95** | 0.80 | 0.54 | 0.80 | **0.87** | **0.87** |
| **acenaphthylene** | | | **0.92** | **0.91** | **0.96** | **0.95** | **0.97** | **0.95** | **0.97** | **0.96** | **0.94** | **0.97** | **0.95** | **0.96** | **0.81** | **0.80** | **0.95** | 0.74 | **0.94** | **0.83** | **0.87** |
| **anthanthrene** | | | | **0.82** | **0.91** | **0.90** | **0.92** | **0.91** | **0.92** | **0.93** | **0.91** | **0.88** | **0.92** | **0.91** | 0.77 | 0.77 | **0.88** | 0.67 | **0.89** | 0.74 | **0.82** |
| **anthracene** | | | | | **0.90** | **0.94** | **0.91** | **0.89** | **0.89** | **0.88** | **0.85** | **0.93** | **0.90** | **0.89** | **0.90** | **0.95** | **0.90** | 0.74 | **0.90** | **0.94** | **0.94** |
| **benzo(b+k)fluoranthene** | | | | | | **0.98** | **0.99** | **1.00** | **0.99** | **0.99** | **0.96** | **0.98** | **0.99** | **0.99** | **0.85** | **0.82** | **0.99** | 0.72 | **0.96** | **0.82** | **0.89** |
| **benzo(a)anthracene** | | | | | | | **0.98** | **0.98** | **0.97** | **0.97** | **0.93** | **0.99** | **0.99** | **0.98** | **0.89** | **0.88** | **0.98** | 0.77 | **0.96** | **0.87** | **0.93** |
| **benzo(a)pyrene** | | | | | | | | **0.99** | **0.99** | **0.99** | **0.96** | **0.98** | **0.99** | **0.99** | **0.82** | **0.81** | **0.98** | 0.74 | **0.98** | **0.80** | **0.87** |
| **benzo(b)fluoranthene** | | | | | | | | | **0.98** | **0.98** | **0.96** | **0.98** | **1.00** | **1.00** | **0.85** | **0.81** | **0.99** | 0.74 | **0.96** | **0.82** | **0.89** |
| **benzo(e)pyrene** | | | | | | | | | | **1.00** | **0.96** | **0.97** | **0.98** | **0.99** | **0.82** | 0.80 | **0.98** | 0.71 | **0.94** | 0.79 | **0.87** |
| **benzo(ghi)perylene** | | | | | | | | | | | **0.96** | **0.96** | **0.98** | **0.98** | **0.80** | 0.78 | **0.98** | 0.71 | **0.95** | 0.78 | **0.86** |
| **benzo(k)fluoranthene** | | | | | | | | | | | | **0.95** | **0.96** | **0.96** | 0.78 | 0.77 | **0.95** | 0.68 | **0.95** | 0.78 | **0.82** |
| **chrysene** | | | | | | | | | | | | | **0.98** | **0.98** | **0.88** | **0.85** | **0.98** | 0.76 | **0.96** | **0.87** | **0.92** |
| **coronene** | | | | | | | | | | | | | | **1.00** | **0.86** | **0.84** | **0.99** | 0.74 | **0.97** | **0.83** | **0.90** |
| **dibenzo(ah)anthracene** | | | | | | | | | | | | | | | **0.85** | **0.81** | **0.99** | 0.74 | **0.96** | **0.82** | **0.89** |
| **fluoranthene** | | | | | | | | | | | | | | | | **0.94** | **0.84** | 0.66 | **0.82** | **0.96** | **0.98** |
| **fluorene** | | | | | | | | | | | | | | | | | **0.82** | 0.62 | **0.83** | **0.95** | **0.95** |
| **indeno(1,2,3-cd)pyrene** | | | | | | | | | | | | | | | | | | 0.70 | **0.95** | **0.83** | **0.88** |
| **naphthalene** | | | | | | | | | | | | | | | | | | | 0.72 | 0.68 | 0.72 |
| **perylene** | | | | | | | | | | | | | | | | | | | | **0.81** | **0.85** |
| **phenanthrene** | | | | | | | | | | | | | | | | | | | | | **0.97** |
| **pyrene** | | | | | | | | | | | | | | | | | | | | | |

## *Comparison between Leigh Marshes and Hadleigh Marsh*

The statistical comparison (Mann-Whitney U tests) of PAH concentrations in Leigh Marshes and Hadleigh Marsh landfills is summarised in Table 3.41. Leigh Marshes landfill contains 179-1840% higher concentrations (median values) of all of the PAHs in its matrix material compared to Hadleigh Marsh landfill, except

Cyclopenta(cd)pyrene which had median concentrations that were below detection in both sites.

**Table 3.41: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in organic contaminant concentrations in the matrix material between Hadleigh Marsh (HM) and Leigh Marshes (LM) landfill sites (critical p value = 0.05)**

| | Mann-Whitney U p value | Significant difference | HM median (n=20) (µg $kg^{-1}$) | LM median (n=24) (µg $kg^{-1}$) |
|---|---|---|---|---|
| acenaphthene | 0.002 | HM<LM | 30 | 142 |
| acenaphthylene | 0.010 | HM<LM | 44 | 95 |
| anthanthrene | <0.001 | HM<LM | 156 | 479 |
| anthracene | <0.001 | HM<LM | 186 | 606 |
| benzo(b+k)fluoranthene | 0.001 | HM<LM | 1435 | 3755 |
| benzo(a)anthracene | <0.001 | HM<LM | 743 | 2010 |
| benzo(a)pyrene | 0.001 | HM<LM | 761 | 2080 |
| benzo(b)fluoranthene | 0.001 | HM<LM | 1055 | 2910 |
| benzo(e)pyrene | 0.001 | HM<LM | 567 | 1575 |
| benzo(ghi)perylene | 0.001 | HM<LM | 577 | 1570 |
| benzo(k)fluoranthene | 0.006 | HM<LM | 407 | 930 |
| chrysene | 0.001 | HM<LM | 771 | 2040 |
| coronene | 0.001 | HM<LM | 144 | 386 |
| cyclopenta(cd)pyrene | Insufficient data | | <10 | <10 |
| dibenzo(ah)anthracene | 0.001 | HM<LM | 150 | 451 |
| fluoranthene | 0.001 | HM<LM | 2050 | 4405 |
| fluorene | <0.001 | HM<LM | 54 | 291 |
| indeno(1,2,3-cd)pyrene | 0.011 | HM<LM | 853 | 1885 |
| naphthalene | <0.001 | HM<LM | 100 | 1840 |
| perylene | 0.001 | HM<LM | 211 | 466 |
| phenanthrene | <0.001 | HM<LM | 622 | 2770 |
| pyrene | 0.003 | HM<LM | 1940 | 3475 |

## 3.10 Total contaminant loads in the landfills

Calculated total contaminant loads for Leigh Marshes for metals and PAHs that are included in the Canadian sediment quality guidelines are shown in Table 3.42 and Table 3.43 respectively. Calculated total contaminant loads for Hadleigh Marsh for metals and PAHs that are included in the Canadian sediment quality guidelines are shown in Table 3.44 and Table 3.45 respectively.

**Table 3.42: Total metal loads in Leigh Marshes landfill's matrix material (range of values determined using lowest and highest typical landfill densities in the calculations)**

| | Matrix material | | Total |
|---|---|---|---|
| Proportion of the site by volume | fines ~44% | putrescible materials ~24% | ~68% |
| Dry mass of material (tonnes) | 173,500-231,300 | 30,850-41,140 | 204,350-272,450 |
| Cd (kg) | 718-958 | 128-170 | 846-1128 |
| Cr (kg) | 11103-14804 | 1975-2633 | 13078-17437 |
| Cu (kg) | 104959-139945 | 18669-24891 | 123627-164837 |
| Pb (kg) | 231083-308111 | 41102-54802 | 272185-362913 |
| Zn (kg) | 267168-356224 | 47520-63360 | 314688-419584 |

**Table 3.43: Total PAH loads in Leigh Marshes landfill's matrix material (range of values determined using lowest and highest typical landfill densities in the calculations)**

| | Matrix material | | Total |
|---|---|---|---|
| Proportion of the site by volume | fines ~44% | putrescible materials ~24% | ~68% |
| Dry mass of material (tonnes) | 173,500-231,300 | 30,850-41,140 | 204,350-272,450 |
| Acenaphthene (kg) | 25-33 | 4.4-5.8 | 29-39 |
| Acenaphthylene (kg) | 16-22 | 2.9-3.9 | 19-26 |
| Anthracene (kg) | 105-140 | 19-25 | 124-165 |
| Benzo(a)anthracene (kg) | 349-465 | 62-83 | 411-548 |
| Benzo(a)pyrene (kg) | 361-481 | 64-86 | 425-567 |
| Chrysene (kg) | 354-472 | 63-84 | 417-556 |
| Dibenzo(ah)anthracene (kg) | 78-104 | 14-19 | 92-123 |
| Fluoranthene (kg) | 764-1019 | 136-181 | 900-1200 |
| Fluorene (kg) | 50-67 | 9-12 | 59-79 |
| Naphthalene (kg) | 319-426 | 57-76 | 376-501 |
| Phenanthrene (kg) | 481-641 | 85-114 | 566-755 |
| Pyrene (kg) | 603-804 | 107-143 | 710-947 |
| Total PAHs* (kg) | 5926-7902 | 1054-1405 | 6981-9307 |
| *including PAHs not in SQGs (anthanthrene, benzo(b+k)fluoranthene, benzo(b)fluoranthene, benzo(e)pyrene, benzo(ghi)perylene, benzo(k)fluoranthene, coronene, indeno(1,2,3-cd)pyrene, perylene) | | | |

**Table 3.44: Total metal loads in Hadleigh Marsh landfill by material type (range of values determined using lowest and highest typical landfill densities in the calculations)**

| | Matrix material | | | Paper | Textiles | Total |
|---|---|---|---|---|---|---|
| Proportion of the site by volume | fines ~16% | putrescible materials ~22% | fines+ putrescible | ~35% | ~4% | ~77% |
| Dry mass of material (tonnes) | 40,060-53,420 | 17,420-23,230 | 57,480-76,650 | 86,940-115,920 | 9,720-12,960 | 154,140-205,530 |
| Cd (kg) | 47-63 | 21-27 | 68-91 | 22-30 | 4-5 | 94-125 |
| Cr (kg) | 2368-3158 | 1030-1373 | 3398-4531 | 810-1080 | 184-246 | 4393-5857 |
| Cu (kg) | 1334-1779 | 580-773 | 1914-2552 | 1121-1494 | 991-1321 | 4026-5368 |
| Pb (kg) | 2525-3366 | 1098-1464 | 3622-4830 | 1620-2159 | 1691-2254 | 6933-9244 |
| Zn (kg) | 8389-11186 | 3647-4863 | 12037-16049 | 14636-19515 | 5075-6767 | 31748-42331 |

**Table 3.45: Total PAH loads in Hadleigh Marsh landfill's matrix material (range of values determined using lowest and highest typical landfill densities in the calculations)**

| | Matrix material | | Total |
|---|---|---|---|
| Proportion of the site by volume | fines ~16% | putrescible materials ~22% | ~38% |
| Dry mass of material (tonnes) | 40,060-53,420 | 17,420-23,230 | 57,480-76,650 |
| Acenaphthene (kg) | 1.2-1.6 | 0.5-0.7 | 1.7-2.3 |
| Acenaphthylene (kg) | 1.8-2.4 | 0.8-1 | 2.5-3.4 |
| Anthracene (kg) | 7.5-9.9 | 3.2-4.3 | 11-14 |
| Benzo(a)anthracene (kg) | 30-40 | 13-17 | 43-57 |
| Benzo(a)pyrene (kg) | 30-41 | 13-18 | 44-58 |
| Chrysene (kg) | 31-41 | 13-18 | 44-59 |
| Dibenzo(ah)anthracene (kg) | 6-8 | 2.6-3.5 | 8.6-11.5 |
| Fluoranthene (kg) | 82-110 | 36-48 | 118-157 |
| Fluorene (kg) | 2.2-2.9 | 0.9-1.3 | 3.1-4.1 |
| Naphthalene (kg) | 4-5.3 | 1.7-2.3 | 5.7-7.7 |
| Phenanthrene (kg) | 25-33 | 11-14 | 36-48 |
| Pyrene (kg) | 78-104 | 34-45 | 112-149 |
| Total PAHs* (kg) | 515-687 | 224-299 | 739-986 |
| *including PAHs not in SQGs (anthanthrene, benzo(b+k)fluoranthene, benzo(b)fluoranthene, benzo(e)pyrene, benzo(ghi)perylene, benzo(k)fluoranthene, coronene, indeno(1,2,3-cd)pyrene, perylene) | | | |

## 3.11 Determining the maximum probable breach extent using geophysical site investigations

The Geonics EM31 is an electromagnetic conductivity meter that detects secondary electromagnetic fields that are induced in conductive media by the electromagnetic field it transmits (Cassidy et al., 2014). It has an operating range of approximately 6 metres in depth, but is most sensitive to objects buried up to 1 metre deep (NcNeill, 1980b; Geovision Geophysical Services, nd). It records two measurements: ground conductivity (quad-phase) and magnetic susceptibility (In-phase) (Geonics Limited, 2013). The ground conductivity measurement is reported as apparent conductivity in mS $m^{-1}$ (Geonics Limited, 2013) and responds to changes in moisture, salinity, soil type (due to differences in porosity, pore water volume, dissolved electrolytes, and amount and composition of colloids) and the presence of non-metallic bulk wastes (NcNeill, 1980a; Geovision Geophysical Services, nd). The In-phase measurement is reported as the ratio of the secondary to primary magnetic field in parts per thousand (Geonics Limited, 2013), and typically only fluctuates where metallic objects are present. Small metallic objects cause a negative measurement over the object and low amplitude positive responses to the sides of it (Geovision Geophysical Services, nd; Reynolds, 2011). If long, linear conductors, e.g. pipelines, are parallel to the EM31 boom (which houses the electromagnetic transmitter and receiver) then a strong positive measurement is usually recorded (Geovision Geophysical Services, nd).

The outputs from the EM31 survey carried out along the length of Hadleigh Marsh landfill embankment are shown overlaid onto aerial photography of the area and compared to a historical map showing known locations of leachate seepage in Figure 3.19. Areas of high conductivity in relation to background levels (purple and red areas in Figure 3.19) may indicate changes in soil type, water content or water quality (Nobes et al., 2000). The entire site is known to be capped with clay taken from the adjacent borrow ditch, it is therefore probable that the areas of high conductivity are related to changes in water content or quality not soil type, which in turn suggests that the high conductivity locations indicate the presence of leachates. Historical Environment Agency records of leachate seepage locations correspond with many of the high

conductivity areas (Environment Agency, 1996), suggesting that leachates have been seeping from the same locations for at least 20 years. The outputs from the transections across Hadleigh Marsh landfill embankment consistently show the highest conductivity at the lowest points of the embankment, which was expected as there is water to both sides – the estuary to the south and the borrow ditch to the north.

Changes in the In-phase signal can be used to indicate the likely location of metallic objects (Reynolds International, 2011), the In-phase signal only fluctuated three times along the length of the embankment: at location 1 historical maps indicate it is likely there is a culvert present that was sealed off when the landfill was constructed (Ordnance Survey Office, 1895), at location 2 it is due to the presence of a metal gate, and at location 3 it is likely to be due to reinforcing bars in the concrete parking bay at that location.

The EM31 survey did not provide any indication of significant variations in the waste mixture, waste depth or of the locations of the subterranean division of Hadleigh Marsh landfill site into waste cells. Hence, it was not possible to determine the locations of the waste cell boundaries. However, the surveys provided data on likely leachate seepage locations along Hadleigh Marsh embankment - the potential for pollution from leachate entering the estuary is considered in the next chapter. Excessive seepage of water through embankments can result in their failure (Defra and Environment Agency, 2007b); therefore, the likely seepage locations may indicate the areas of the embankment that have the greatest potential to breach and would provide a useful input into any future breach modelling for the site.

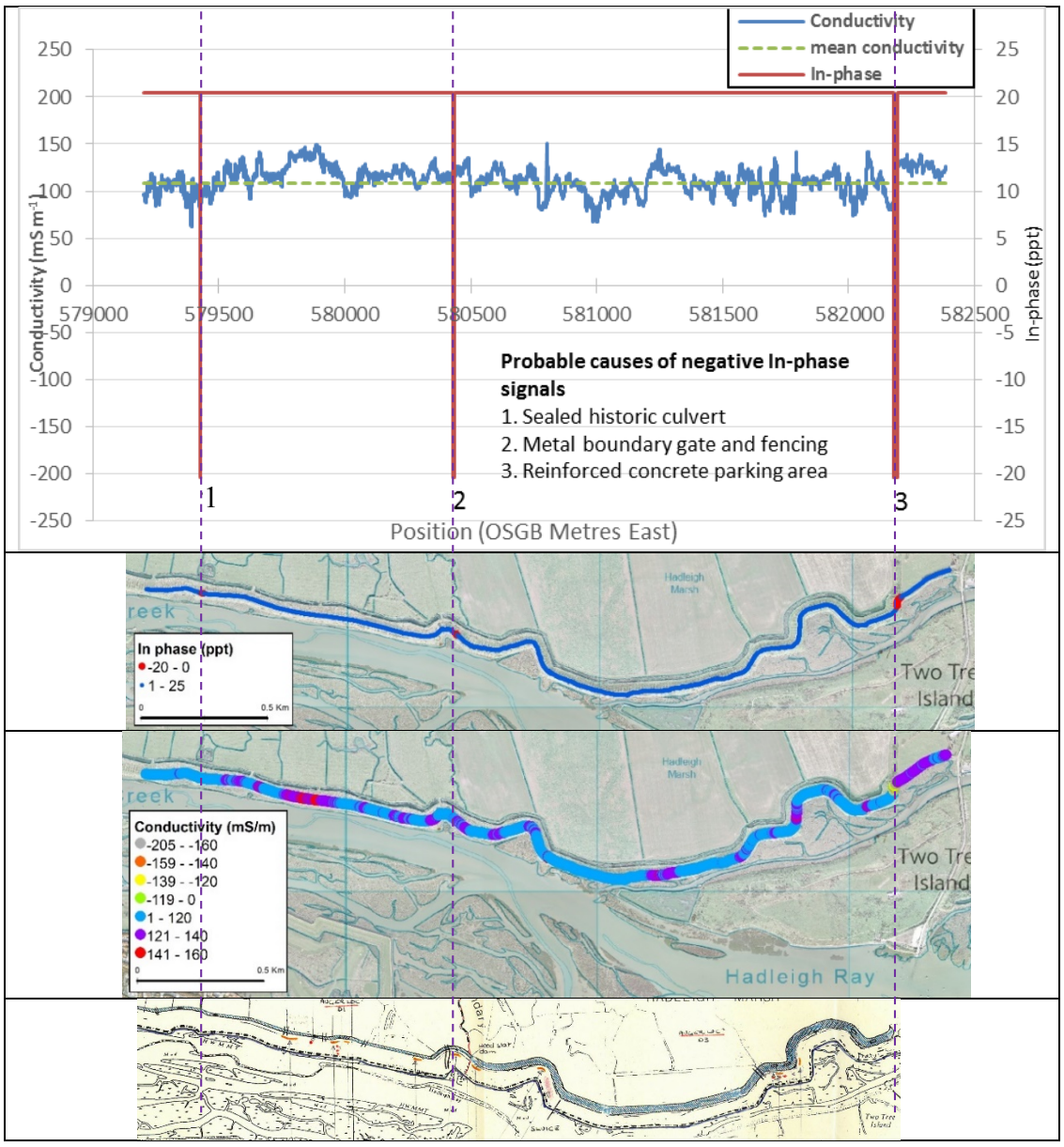



**Figure 3.19: EM31 survey results (along embankment) and known locations of historical leachate seepage (orange lines, bottom map) from EA records (Environment Agency, 1996, Appendix A14.1) (Contains Ordnance Survey data © Crown copyright and database right 2014, © GeoPerspectives). (North up the page)**

## 3.12 Contaminant magnitude and variability

### 3.12.1 Contaminant concentrations in the waste compared to previous studies

There have been few published studies of contaminants in solid waste in historic landfills. The results of the matrix material analyses for inorganic contaminants from this research are compared to the results of the previous studies in Table 3.46. No equivalent studies were found for organic contaminants. The analysis methods used by the previous studies are not directly comparable with the methods used for this research, e.g. Quaghebeur *et al*. (2013) and Zhao *et al*. (2007) used hydrofluoric acid digests, which, depending on the mineral structure of the material being analysed, can increase metal extraction by more than tenfold compared to the methods used by this research (Chen and Ma, 2001). Nevertheless, with the exception of Cu in Hadleigh Marsh, the contaminant concentrations in the matrix materials from Leigh Marshes and Hadleigh Marsh are in the same range and/or of the same order of magnitude as those found in previous studies. However, this reflects the wide variety of results obtained in previous studies rather than any consistency in contaminant concentrations between sites (see Table 3.46).

The results of the Hadleigh Marsh paper analyses for inorganic contaminants are compared to the results of a previous study (Quaghebeur et al., 2013) of mixed paper and cardboard samples from excavated landfill waste in Table 3.47. Metal concentrations in the Hadleigh Marsh samples are considerably lower than in the previous study, this may be related to the extraction techniques used, differences in the species of wood used for manufacturing paper in different countries (Sakthivel et al., 2012) or differences in inks, which vary in metal content by colour and manufacturing era (Telschow, 1994; Canadian Printing Ink Manufacturers' Association, 2010). The higher metal concentrations in the paper excavated from other landfill sites may also be related to metal sorption to the paper from leachate that has transported metals from other waste materials within those sites.

**Table 3.46: Inorganic contaminant concentrations in Leigh Marshes and Hadleigh Marsh matrix material compared to previous studies (underlined = at least one order of magnitude lower than previous studies)**

| | | Ca | Cd | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| **This research (mg kg$^{-1}$)** | | | | | | | | | | |
| **Leigh Marshes matrix** | | 24553 | 4.14 | 64 | 605 | 114813 | 870 | 141 | 1332 | 1540 |
| **Hadleigh Marsh matrix** | | 11116 | 1.18 | 59 | 33 | 48533 | 422 | 45 | 63 | 209 |
| **Previous research (mg kg$^{-1}$)** | | | | | | | | | | |
| **Belgium MSW 'soil', dates landfilled** | **1980-1985** | 31000 | 8.5 | 770 | 285 | 34000 | | 335 | 500 | 670 |
| | **1985-1990** | 35000 | 8.4 | 380 | 205 | 39000 | | 164 | 310 | 735 |
| | **1990-1995** | 33000 | 3.3 | 720 | 760 | 18000 | | 160 | 180 | 800 |
| | **1995-2000** | 14000 | 3.3 | 113 | 107 | 17000 | | 46 | 172 | 463 |
| **Belgium IW 'soil', dates landfilled** | **1985-1990** | 73000 | 15 | 3800 | 1200 | 53000 | | 2200 | 1100 | 2600 |
| | **1995-2000** | 78000 | 19 | 5730 | 5750 | 54000 | | 4640 | 2640 | 5600 |
| **New Jersey, USA 'fines'** | | | 1.2 | 24 | | | | | 46 | 406 |
| **South China mixed waste, age** | **2 years** | | 10.7 | 120.6 | | | | | 171.3 | 556.2 |
| | **4 years** | | 1.1 | 73.5 | | | | | 134.2 | 549.4 |
| | **6 years** | | 2.7 | 174.6 | | | | | 141 | 602.1 |
| | **8 years** | | 6.6 | 252.1 | | | | | 372.2 | 652.9 |
| | **10 years** | | 5.6 | 169.3 | | | | | 77.9 | 634.2 |
| **Thailand <25 mm fraction** | | | 4.2 | 166.6 | 2245 | | 947 | 47.8 | 132 | 1497 |

Sources: Belgium data from Quaghebeur et al. (2013), New Jersey data from Hull et al. (2005), South China data from Zhao et al. (2007) and Thailand data from Prechthai et al. (2008)
MSW=Municipal Solid Waste, IW=Industrial Waste

**Table 3.47: Inorganic contaminant concentrations in Hadleigh Marsh paper samples compared to a previous study of mixed paper and cardboard samples in Belgium (underlined = at least one order of magnitude lower than previous studies)**

| | | Ba | Ca | Cd | Cr | Cu | Fe | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| **This research (mg kg$^{-1}$)** | | | | | | | | | | |
| **Hadleigh Marsh paper** | | 24 | 4589 | 0.25 | 9.32 | 13 | 2530 | 4.93 | 19 | 168 |
| **Previous research (mg kg$^{-1}$)** | | | | | | | | | | |
| **Belgium MSW paper/ cardboard, dates landfilled** | **1980-1985** | 240 | 17000 | 1.4 | 310 | 150 | 26000 | 120 | 440 | 900 |
| | **1985-1990** | 480 | 32000 | 16 | 200 | 210 | 29000 | 56 | 330 | 560 |
| | **1990-1995** | 180 | 27000 | <0.40 | 140 | 570 | 6500 | 45 | 54 | 520 |
| | **1995-2000** | 280 | 33000 | 2.7 | 160 | 100 | 30000 | 86 | 380 | 1900 |

Sources: Belgium data from Quaghebeur et al. (2013)

No previous studies of contaminants in wood or textiles in historic landfill sites were found, but there have been studies of inorganic contaminants in wood and textiles in freshly collected household waste (Riber et al., 2009; Eisted and Christensen, 2011). No equivalent studies were found for organic contaminants. The results of the Hadleigh Marsh wood and textile sample analyses from this research are compared to the results of the previous studies in Table 3.48.

The analysis methods used by the previous studies (hydrofluoric acid digests) are not directly comparable with the methods used for this research and could be expected to report higher metal concentrations. However, the Hadleigh Marsh wood samples had concentrations of Fe, Ni, Pb and Zn at least one order of magnitude higher and concentrations of B, Co and K at least one order of magnitude lower than the wood in fresh waste. No correlations were found between these metals and moisture content in the Hadleigh Marsh wood samples, so it is unlikely these metal concentrations have been influenced by leachate movement through the site. Different species of wood contain Fe, K, Mg, Na, Ni, Pb and Zn in varying concentrations (Sakthivel et al., 2012) and B, Co, Fe, K, Mg, Na, Ni, Pb and Zn are currently or have historically been used in wood paints and preservatives, and fire retardant coatings (Nicholas, 1973; Mahltig et al., 2008; Freeman et al., 2009; Pereyra and Giudice, 2009; Hansen et al., 2015; Airedale Chemical, 2016). Therefore, the wide variety of wood, wood paints and preservatives and fire retardant chemicals may account for the differences in metal concentrations between the modern wood waste and wood from Hadleigh Marsh waste. In addition, changes in the composition of wood paints and preservatives have occurred as some metals have been banned in their use due to health concerns, e.g. Pb removal from consumer paints in the 1980s.

The Hadleigh Marsh textile samples had concentrations of Pb and Zn at least one order of magnitude higher than the textiles in fresh waste. No correlation was found between moisture content and Pb or Zn in the Hadleigh marsh textiles, which means the elevated concentrations of Pb and Zn are unlikely to be related to leachate movement within Hadleigh Marsh. The elevated concentrations of Pb and Zn may be due to the presence of dirt acquired by the textiles during landfilling or during their excavation.

**Table 3.48: Inorganic contaminant concentrations in Hadleigh Marsh wood and textile samples compared to studies of fresh waste (grey cells=at least one order of magnitude higher than previous studies, underlined=at least one order of magnitude lower than previous studies)**

| | Wood (mg kg$^{-1}$) | | Textiles (mg kg$^{-1}$) | | |
|---|---|---|---|---|---|
| | **Hadleigh Marsh** | **Greenland** | **Hadleigh Marsh** | **Germany** | **Denmark** |
| **Ag** | <0.06 | <0.08 | 0.49 | | |
| **Al** | 1299 | 1150 | 4051 | | |
| **B** | <u>≤5.83</u> | 128 | <5.83 | | |
| **Ba** | 42 | 39.7 | 87 | | |
| **Ca** | 3880 | 3350 | 31507 | | |
| **Cd** | <0.72 | 0.128 | <0.72 | 0.045 | 0.015 |
| **Co** | <u>≤0.31</u> | 38.7 | <0.31 | | |
| **Cr** | 10 | 10 | 19 | | |
| **Cu** | 36 | 32 | 102 | | |
| **Fe** | 13096 | 747 | 13847 | | |
| **K** | 535 | 964 | 676 | | |
| **Li** | <0.03 | <0.2 | <0.03 | | |
| **Mg** | 703 | 1120 | 1100 | | |
| **Mn** | 84 | 93.9 | 273 | | |
| **Na** | 451 | 1360 | 264 | | |
| **Ni** | 13 | 2.51 | 17 | | |
| **Pb** | 140 | 31.7 | 174 | 1.56 | 4.69 |
| **Sr** | 18 | 20.9 | 123 | | |
| **Zn** | 1094 | 118 | 522 | 23.1 | 6.65 |
| Sources: Greenland (Eisted and Christensen, 2011), Germany and Denmark (Riber et al., 2009). | | | | | |

### 3.12.2 Contaminant variability at a local scale (individual trial pits)

Contaminant concentrations in the sites were found to be highly variable at the local scale, i.e. within individual trial pits. Each Leigh Marshes trial pit sampled waste from a volume of less than 1 $m^3$, but in the matrix material there were differences of up to two orders of magnitude between the minimum and maximum concentrations of each of the trace metals, less than one order of magnitude difference in the minimum and maximum concentrations of each of the major metals, and up to three orders of magnitude between the minimum and maximum concentrations of each of the PAHs in each trial pit. Each Hadleigh Marsh trial pit sampled waste from a volume of less than 2 $m^3$, but in the matrix material there were differences of up to four orders of magnitude between the

minimum and maximum concentrations of each of the trace metals, up to an order of magnitude between the minimum and maximum concentrations of each of the major metals, and up to two orders of magnitude between the minimum and maximum concentrations of each of the PAHs in each trial pit. Only Hadleigh Marsh contained sufficient wood, paper and textiles to enable their analysis, and these materials were only suitable for analyses to determine metals not PAHs. In the wood samples there were differences of up to four orders of magnitude between the minimum and maximum concentrations of each of the trace metals, up to three orders of magnitude between the minimum and maximum concentrations of each of the major metals in individual trial pits. In the paper samples there were differences of up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, up to three order of magnitude between the minimum and maximum concentrations of each of the major metals in individual trial pits. In the textile samples there were differences of up to three orders of magnitude between the minimum and maximum concentrations of each of the trace metals, up to two order of magnitude between the minimum and maximum concentrations of each of the major metals in individual trial pits.

The high variability in contaminant concentrations reflects the highly heterogeneous nature of waste materials in the landfill sites and the high variability of contaminants in contaminated land in general (Allen, 2001; Blight and Fourie, 2005; Taylor and Ramsey, 2006). No previous studies of solid waste excavated from historic landfills have published contaminant variability data for comparison, but contaminant variability in the waste samples excavated for this research is comparable to that found in a study of mixed material samples of modern day waste from households in Denmark, which showed differences of up to six orders of magnitude between the minimum and maximum concentrations of trace metals and up to five orders of magnitude between the minimum and maximum concentrations of major metals (Riber et al., 2007). No equivalent studies were found for PAHs.

### 3.12.3 Contaminant variability within sites

Contaminant concentrations in each material type show the same level of variability when considered for the entirety of each landfill site as they do for individual trial pits, with the exception of the major metals in the Leigh Marshes matrix material for which there is a greater difference between the minimum and maximum concentrations (up to

one order of magnitude) when the site is considered as a whole. In addition, there were significant differences in metal and PAH concentrations between the two trial pits within each of the landfill sites.

The high heterogeneity of environmental conditions in landfills can result in differences in leachate movement through different areas of a landfill causing uneven migration of contaminants (either dissolved or bound to suspended solids) (Alesii et al., 1980; Bendz et al., 1997; Allen, 2001; van der Sloot et al., 2003). Leachate flow rates and the transport of soluble contaminants increase with increased moisture content in waste (Imhoff et al., 2007). Therefore, differences in contaminant concentrations between different areas of the landfill sites may be related to differences in leachate flow through those areas if they contain significantly different moisture contents. If leachate movement has transported contaminants from a source elsewhere in the waste to the material sampled, then positive correlations between contaminant concentrations and the moisture content of that material can be expected. Conversely, if contaminants have been transported away from the material sampled by leachate movement, then negative correlations between their concentrations and that material's moisture content can be expected.

In the Leigh Marshes matrix material, differences in Ag, Cd, Cr, Fe and Zn concentrations between the two trial pits can be explained by leachate movement as their concentrations in the matrix material are positively correlated with moisture content and moisture content was significantly higher in the trial pit which contained the greatest concentrations of these metals. Differences in Al, Ca, Co, K, Li, Mn, Na, Ni and Sr concentrations between the two trial pits can also be explained by leachate movement as their concentrations in the matrix material are negatively correlated with moisture content and moisture content was significantly higher in the trial pit that contain the lowest concentrations of these metals. Differences in acenaphthene, fluorene, naphthalene, B, Ba and Pb concentrations between the two trial pits cannot be explained by leachate movement as the concentrations of these contaminants in the matrix material showed no correlation with moisture content.

In Hadleigh Marsh, there was no significant difference in moisture content in the matrix material between the two trial pits and no significant difference in moisture content in the wood between the two trial pits. Therefore, the differences in PAH, B, Ca, Cd, Co,

Cu, Cr, K, Li, Mg, Mn, Na, Pb, Sr and Zn concentrations in the matrix material and Ca, Cr, Fe, Na, Pb and Sr concentrations in the wood samples between the two trial pits cannot be explained by differences in leachate movement.

In the Hadleigh Marsh paper samples, differences in Al concentrations between the two trial pits can be explained by leachate movement as Al in the paper is negatively correlated with moisture content and moisture content was significantly higher in paper in the trial pit that contained the lowest concentrations of Al in paper. Differences in Ba, Ca, Cr, Fe, Mn, Na, Sr and Zn concentrations in paper between the two trial pits cannot be explained by leachate movement as the concentrations of these contaminants showed no correlations with moisture content in paper.

In the Hadleigh Marsh textile samples, differences in Ag, Cu and Fe concentrations between the two trial pits can be explained by leachate movement as their concentrations in the textile samples are positively correlated with moisture content and moisture content was significantly higher in textiles in the trial pit which contained the greatest concentrations of these metals. Differences in Ca and Sr concentrations between the two trial pits can also be explained by leachate movement as their concentrations in the textile samples are negatively correlated with moisture content and moisture content was significantly higher in textiles in the trial pit that contain the lowest concentrations of these metals. Differences in B, Ba, Co, K, Mg, Mn, Na, Pb, and Zn concentrations in textiles between the two trial pits cannot be explained by leachate movement as the concentrations of these contaminants showed no correlation with moisture content in textiles.

The differences between contaminant concentrations in different trial pits that cannot be explained by leachate movement reflect the fact that there is a high heterogeneity of waste materials in landfill sites, due to changes in the types of materials landfilled with time, differences in age and degradation of waste, and poor mixing of waste during construction, (Alesii et al., 1980; Noble et al., 1989; Bagchi, 1994; Bendz et al., 1997; Allen, 2001; van der Sloot et al., 2003; Blight and Fourie, 2005; Sormunen et al., 2008). In addition, even materials nominally of the same type and from the same era may contain significantly different concentrations of contaminants, e.g. PAHs in wood ash and coal ash depend upon combustion conditions (Tsibart and Gennadiev, 2013).

As well as the differences between contaminant concentrations in the two trial pits in this research, there were significant differences between some PAH and metal concentrations measured in this research and Halcrow's previous site investigations (Halcrow Group Ltd, 2012). These differences further demonstrate the very heterogeneous nature of the waste and the difficulty in ensuring that sampling of a landfill is representative of the entire site. However, the differences in median contaminant concentrations were less than one order of magnitude, except for naphthalene in Leigh Marshes, which was one order of magnitude higher in the results of this research. For Cd the differences between concentrations measured in this research and Halcrow's research may be attributable to this research using a more sensitive analytical method, i.e. ICP-MS, as Cd concentrations in 40% of Halcrow's Leigh Marshes samples and in 60% of Halcrow's Hadleigh Marsh samples were below the limit of detection of their analysis method. No moisture content data are available for the Halcrow data, so the possible relationship between leachate movement and differences in the contaminant concentrations measured in the two studies cannot be investigated.

### 3.12.4 Waste material and contaminant variability between sites

There were significant differences in the physical appearance of the waste materials in the two landfill sites. The waste extracted from Leigh Marshes landfill was predominantly composed of a fine-grained particulate matrix interspersed with broken bricks, glass and ceramics, and small quantities of paper, rubber, bones, plant materials and wood. The matrix material is likely to consist of ash/dust/screenings and degraded putrescible materials, as these formed the majority of MSW materials during the period of time that this site was in use (see Figure 3.2). However, it may also contain materials from commercial and industrial sources as the site is known to have received household, commercial and industrial wastes (Environment Agency, 2015b). No information exists relating to the types of materials present in commercial or industrial wastes landfilled at Leigh Marshes.

The waste extracted from Hadleigh Marsh landfill had the appearance of present-day waste (before separation for recycling), consisting of plastics, ceramics, textiles (shoes, carpets, clothes), paper, wood, batteries, soil, and putrescible materials.

The change in landfilled materials between sites from different eras is consistent with previous research, particularly the significant increase in plastics and decrease in ash/dust/screenings after the mid-1960s (Bridgwater, 1986; Burnley, 2007a; Parfitt, 2009).

There were significant differences in metal and PAH concentrations in the matrix material between the two study sites. Leigh Marshes landfill contains significantly higher concentrations of all of the PAHs in its matrix material compared to Hadleigh Marsh landfill, except Cyclopenta(cd)pyrene which had median concentrations below detection in both sites. Leigh Marshes landfill contains significantly higher concentrations of Ag, B, Ba, Ca, Cd, Co, Cr, Cu, Fe, Mn, Na, Ni, Pb, Sr and Zn in the matrix material than Hadleigh Marsh landfill. Median concentrations of Ba, Cu and Pb in the matrix material were an order of magnitude higher in Leigh Marshes than in Hadleigh Marsh. Hadleigh Marsh landfill contains significantly higher concentrations of Al, K, Li and Mg in the matrix material than Leigh Marshes landfill.

There were moderate or strong correlations between most of the PAHs in the Leigh Marshes matrix materials, exceptions were acenaphthene, fluorene and naphthalene which had few correlations with other PAHs, this suggests most of the PAHs in Leigh Marshes have a common source. In the Hadleigh Marshes matrix material there were strong correlations between most of the PAHs, exceptions were acenaphthene and naphthalene which predominantly had moderate correlations with other PAHs, this suggests the PAHs in Hadleigh Marsh have a common source. No correlations between moisture content and PAHs were found in either site. Wood ash and coal ash are known to contain high levels of PAHs and are likely to be a primary source of PAHs in both sites (Liu et al., 2008; Tsibart and Gennadiev, 2013). There has been a reduction in ash in household waste since the mid-1960s (Bridgwater, 1986; Burnley, 2007a), which could explain the significantly higher concentrations of PAHs in Leigh Marshes landfill (constructed 1955-1967) compared to Hadleigh Marsh landfill (constructed 1980-1987).

The significantly higher concentrations of most of the metals analysed in the matrix material in Leigh Marshes compared to Hadleigh Marsh may also be related in part to the differences in ash content as ash is known to contain high quantities of most of the metals (Smolka-Danielowska, 2006). However, correlations between these metals are a complex mixture of positive and negative in both sites, suggesting that they are not from

a single source. There are also a mixture of positive and negative correlations between them and moisture content in Leigh Marshes, and positive correlations between them and moisture content in Hadleigh Marsh. Therefore, it is not possible to give a definitive judgement on the number of sources or the reasons for the differences between the sites as leachate movement appears to have influenced the concentrations present in the matrix material at both sites. However, it is likely that the differences between the sites are predominantly due to changes in material types typically being landfilled between the construction periods of the two landfills (Figure 3.2) and the fact that Leigh Marshes contains unknown industrial waste materials in addition to the household and commercial wastes that both sites contain. Industrial waste having higher concentrations of metals than municipal solid waste is consistent with research in Belgium (Table 3.46). As seen with differences within sites, differences between sites also reflect the fact that there is a high heterogeneity of waste materials in landfill sites, due to changes in the types of materials landfilled with time, differences in age and degradation of waste, and poor mixing of waste during construction (Alesii et al., 1980; Noble et al., 1989; Bagchi, 1994; Bendz et al., 1997; Allen, 2001; van der Sloot et al., 2003; Blight and Fourie, 2005; Sormunen et al., 2008).

### 3.12.5 Should generic contaminant datasets be used to represent all historic coastal landfill sites?

This research found that sampling and analysing solid waste materials from different areas of a single landfill site, and different studies of multiple locations within a single landfill site, can result in significantly different contaminant datasets. This is consistent with studies of landfill leachates that concluded intensive sampling is required to characterise landfill sites due to the high heterogeneity of the materials within them (Sormunen et al., 2008) and more general studies of contaminated land that found sampling uncertainty can contribute 90% or more to the overall data uncertainty in contaminated land datasets (Crumbling et al., 2001). As there are significant levels of uncertainty about whether the contaminant datasets from highly heterogeneous contaminated sites are actually representative of the sites they were obtained from, they cannot be assumed to be representative of other sites even if they are from the same era, in the same environmental conditions and contain the same waste streams.

In addition, this research found that there can be significantly different contaminant concentrations between landfill sites with different mixtures of waste types; therefore, the validity of the conclusions of risk assessments that utilise a single generic contaminant dataset to represent all landfill sites is highly questionable (e.g. Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013). This is consistent with landfill leachate studies that concluded that results were not transferrable between sites and that each site must be individually assessed (e.g. Sormunen et al., 2008).

Guidance for contaminated land investigation recommends the use of sampling grids with centres of 10 m or less where heterogeneous contamination is anticipated (British Standards Institution (BSI), 2011). For a site the size of Hadleigh Marsh, this would require circa 2,500 sampling points, which is clearly impracticable even without the consideration of replication or sampling at multiple depths at each location. Even if sampling at such a high resolution were practicable, invasive sampling of that magnitude could seriously harm the structural integrity of the Hadleigh Marsh embankment. The number of samples required can be reduced by over 70% using geostatistical procedures (Verstraete and Van Meirvenne, 2008), but this would still require circa 750 sampling points to investigate Hadleigh Marsh, which would be prohibitively expensive and potentially detrimental to the structural integrity of the site and the protected ecological areas within it. In addition, at that sampling resolution, based on their surface areas, a total of nearly 320,000 sampling points would be required to assess all of the 1264 historic coastal landfills at this resolution to provide data for a national scale risk assessment.

A more practicable approach to localised studies would be to undertake limited sampling and analysis to determine the types of materials present and obtain indicative contaminant datasets. These datasets could potentially be less accurate, but are likely to be within the same order of magnitude as those from more intensive site investigations and therefore provide a reasonable indication of the pollution potential of a site. When undertaking risk assessments on a regional or national scale, even limited sampling of all landfill sites is likely to be prohibitively expensive and time consuming. The site works to obtain waste samples for this research cost circa £3,000 per landfill, and, at the time of writing, analyses costs at the EA National Laboratory were £20 per sample for metal analysis and £35 per sample for PAH analysis. Therefore it would be appropriate

to screen sites to identify those that pose the highest risk based on existing data before deciding which sites require invasive investigations. Such a screening assessment should consider the relative risks of different types of waste present within the sites, i.e. household, commercial, industrial, etc., (after Alaska Department of Environmental Conservation, 2015) rather than assuming all sites contain the same contaminant concentrations and materials. The relative risk of different types of waste is further considered in the development of a screening risk assessment in Chapter 5.

## 3.13 Assessment of the risk posed by erosion of waste from Hadleigh Marsh and Leigh Marshes landfills

It is an offence under the Water Framework Directive to allow materials to enter surface waters if they would cause deterioration by chemically or physically modifying the environment (Council Directive, 2000). To fully assess the impact of eroded waste and determine whether such deterioration would occur, it is necessary to be able to predict the rate of erosion and the total mass of waste released. It is known that landfill instability and the slope failure of elevated landfills are often caused by excessive water infiltrating into the waste and increasing pore pressure (Blight and Fourie, 2005; Peng et al., 2016), but there are currently no suitable modelling tools for assessing the rate of waste erosion due to its complex mechanical properties being poorly understood (Dixon and Jones, 2005).

The worst case scenario, where an entire site catastrophically fails rapidly releasing all of its waste, can be assessed (Neuhold, 2013). Leigh Marshes is estimated to contain circa 800,000 $m^3$ of waste (see section 3.2.1), which could result in approximately 204,500-272,500 tonnes (dry mass) of the matrix material being released, assuming the waste has the typical composition of household waste of the time. If the entire site were to erode, the matrix material could contain, approximately 850-1,130 kg of Cd, 13,080-17,440 kg of Cr, 123,630-164,840 kg of Cu, 272,185-362,915 kg of Pb, and 314,690-419,585 kg of Zn based on median concentrations. To put this into context, the upper figures are equivalent to 638% of the annual input of Cu and 327% of the annual input of Zn into the Thames Estuary from all known sources (Stevenson and Ng, 1999). The matrix material would carry a total PAH load of approximately 6,980-9,310 kg. There is no information available about the construction of the site and whether it is divided into waste cells, which could limit the extent of a failure.

Similarly, catastrophic failure of a site the size of Hadleigh Marsh landfill, which contains circa 500,000 $m^3$ of waste (Essex County Council, n.d.), could result in approximately 57,500-76,600 tonnes (dry mass) of the matrix material, 86,900-115,900 tonnes (dry mass) of paper and 9,700-13,000 tonnes (dry mass) of textiles being released, assuming the waste has the typical composition of household waste of the time. However, Hadleigh Marsh landfill is over 4 km long and records indicate it was constructed of seven waste cells (Environment Agency, 1996), so it is unlikely the

entire embankment would fail at once, it is more probable only a single waste cell would fail.

No data have been found to indicate the size of each waste cell, but if it is assumed that the cells are equally sized, the failure of one could release approximately 8,200-10,900 tonnes (dry mass) of the matrix material, 12,400-16,600 tonnes (dry mass) of paper and 1,400-1,900 tonnes (dry mass) textiles. Based on median concentrations of metals in each material, the matrix material, paper and textiles would between them contain approximately 13-18 kg of Cd, 630-835 kg of Cr, 575-765 kg of Cu, 990-1,320 kg of Pb, and 4,535-6,405 kg of Zn. To put this into context, the upper figures are equivalent to less than 3% of the annual input of Cu and less than 5% of the annual input of Zn into the Thames Estuary from all known sources (Stevenson and Ng, 1999). The matrix material would carry a total PAH load of approximately 105-140 kg.

The waste material has the potential to physically and chemically alter the estuarine environment if eroded, it could increase localised suspended particulate matter concentrations and nutrient loads, reduce dissolved oxygen concentrations, and physically damage benthos in the estuary by crushing them, but there have been no studies of the impact of landfill debris on the marine environment (Pope et al., 2011). In addition, benthos may be smothered when buried by waste as most benthic species require a connection to the sediment-water interface to breathe and feed (Bolam and Rees, 2003). For example, if a single waste cell erodes from Hadleigh Marsh, the total volume of material released would be circa 71,400 $m^3$, which is sufficient material to cover twenty Wembley football pitches to a depth of 0.5 metres. It can take between 1 and 4 years for benthic communities on estuarine beds to recover from the deposition of large volumes of materials (Bolam and Rees, 2003).

Physical harm to fauna in the estuary could also result from the release of plastics from landfill sites, the mechanical impacts of plastics on marine organisms include starvation or suffocation due to entanglement, and injury due to ingestion, which can result in reduced feeding, internal injuries, gastrointestinal blockages and death (Derraik, 2002; Zarfl and Matthies, 2010). It is estimated that plastic litter kills up to one million seabirds and 100,000 marine mammals annually (Cefas, 2004). Plastics can also carry high concentrations of inorganic and organic contaminants (up to two orders of magnitude higher than natural sediments) which increase the longer the plastics have

been exposed to the contaminant source, providing a pathway for these contaminants to be transported vast distances (Mato et al., 2001; Teuten et al., 2009; Ashton et al., 2010; Zarfl and Matthies, 2010; Rochman et al., 2014) and harming organisms when the plastics are ingested (Derraik, 2002). Household waste of Leigh Marshes' era typically had negligible amounts of plastic and none were observed in the samples collected, but the waste from one Hadleigh Marsh cell could include approximately 2,900 to 3,860 tonnes of plastics[ii]. Plastics excavated from the Hadleigh Marsh trial pits were of a wide variety of sizes and types, ranging from small polystyrene balls from packaging materials to plastic sheeting, it is likely that microscopic plastics are also present due to the breakdown of macroscopic plastics (Derraik, 2002). The adverse impacts of microplastics on the marine environment are well documented, these include physical and chemical damage to biota when ingested with the potential for the adverse effects to be transferred through the food chain (e.g. Wright et al., 2013; Browne et al., 2015).

The ecological impacts of contaminated waste materials eroding from landfills would depend upon their toxicity to flora and fauna, which can be assessed using sediment quality guidelines (see below), and the affects upon water quality of contaminants released into solution from the eroded waste, which are investigated in Chapter 4.

**Inorganic contaminant concentrations in the matrix material compared to sediment quality guidelines and background concentrations**

Contaminant concentrations in the matrix material are compared with NOAA guideline background sediment concentrations and Canadian sediment quality guidelines for Cd, Cr, Cu, Pb and Zn in Figure 3.20 (Buchman, 2008; Canadian Council of Ministers of the Environment, 2002). Table 3.49 shows the percentage of samples from each site that exceed the ISQGs and PELs. Also shown for comparison is the percentage of surface sediment samples from a nearby (within 1.5 km) SSSI/Ramsar/SPA saltmarsh that exceed the ISQGs and PELs (sediment datasets from O'Shea, 2016).

The majority of the measured metal concentrations in the matrix material from both sites exceed the ISQGs, the majority of metal concentrations in Leigh Marshes matrix material also exceed the PELs and many of the metal concentrations in Hadleigh Marsh matrix material exceed the PELs. The saltmarsh sediments also contain high

[ii] Plastics ~7.5% of landfill of the era, and assuming moisture content of plastic is zero when landfilled.

concentrations of metals which frequently exceed the sediment quality guidelines, but Cr, Cu, Pb and Zn concentrations in Leigh Marshes matrix material (Table 3.50) and Cr, Cu and Zn concentrations in Hadleigh Marsh matrix material are significantly higher (Table 3.51). No data for Cd in the saltmarsh sediment was available and Pb in the saltmarsh sediment was at comparable concentrations to Hadleigh Marsh matrix material. This suggests that although sediments in the Thames are already moderately contaminated (Attrill and Thomes, 1995), metal concentrations in eroded matrix material could have both an immediate adverse ecological impact locally and a long-term adverse ecological impact wherever the waste is dispersed (subject to any dilution effects). This inference is supported by studies that have shown elevated concentrations of metals in sediments and biota around a landfill site that has already started to erode in Lyme Regis, UK (Pope et al., 2011).

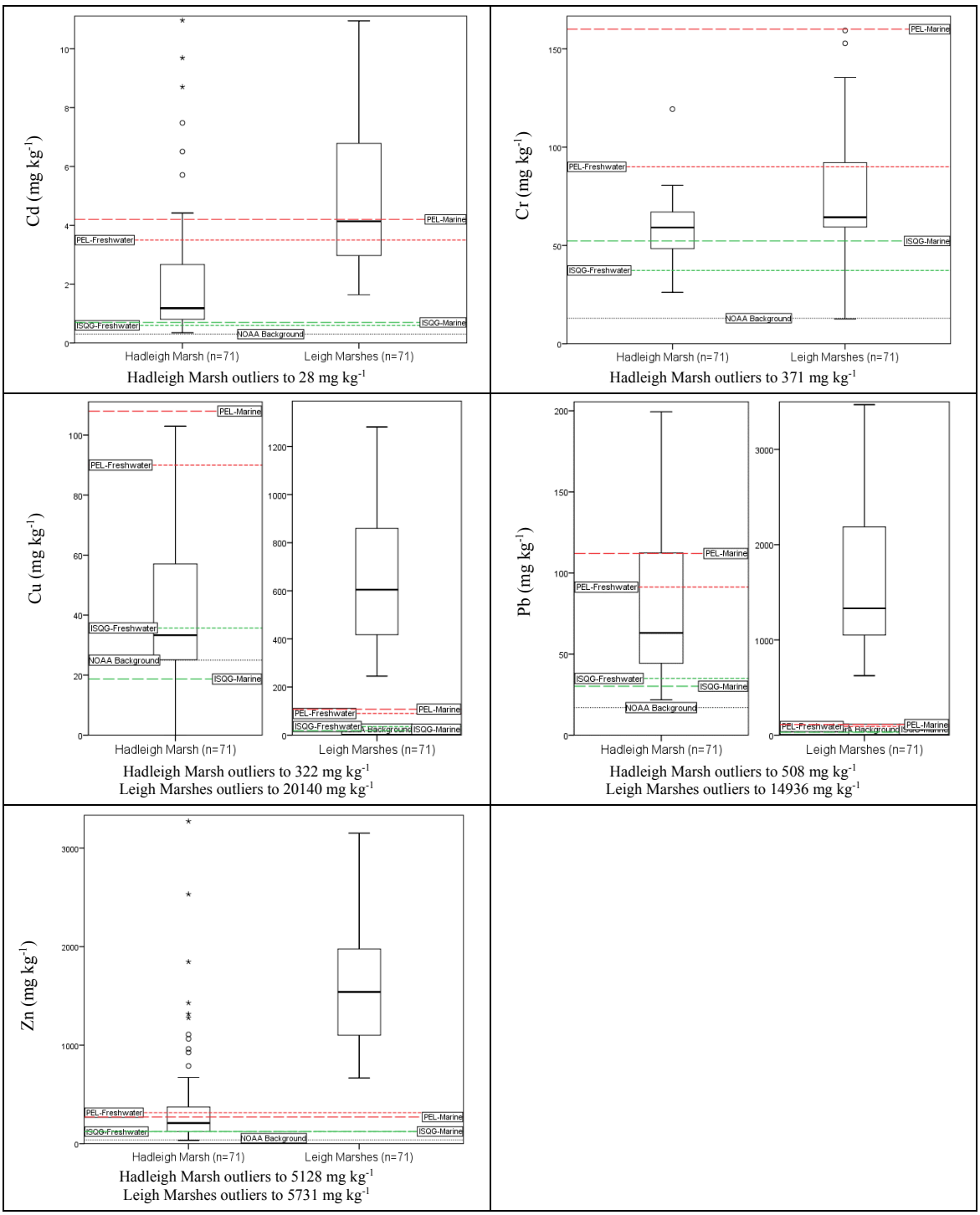



**Figure 3.20: Inorganic contaminant concentrations in the matrix material compared to NOAA background levels for sediment and Canadian sediment quality guidelines (where available)**

**Table 3.49: Percentage of matrix material samples from Leigh Marshes and Hadleigh Marsh landfills and surface sediment samples from a nearby saltmarsh that exceed the Canadian sediment quality guidelines for metals**

| | Leigh Marshes (n=71) | | | | Hadleigh Marsh (n=71) | | | | Saltmarsh (n=91)[1] | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | % >ISQG | | % >PEL | | % >ISQG | | % >PEL | | % >ISQG | | % >PEL | |
| | fresh | marine | fresh | marine | fresh | marine | fresh | marine | fresh | marine | fresh | marine |
| Cd | 100 | 100 | 56 | 48 | 90 | 86 | 23 | 21 | No data | | | |
| Cr | 99 | 93 | 27 | 0 | 96 | 63 | 3 | 1 | 25 | 7 | 0 | 0 |
| Cu | 100 | 100 | 100 | 100 | 46 | 92 | 3 | 7 | 32 | 99 | 2 | 2 |
| Pb | 100 | 100 | 100 | 100 | 88 | 92 | 35 | 25 | 97 | 98 | 16 | 12 |
| Zn | 100 | 100 | 100 | 100 | 75 | 75 | 30 | 37 | 62 | 61 | 2 | 3 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data from O'Shea, 2016, p. 232-233)
NB Canadian guidelines do not specify Interim Sediment Quality Guidelines (ISQGs) or Probable Effect Levels (PELs) for the other metals measured.

**Table 3.50: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between metal concentrations in the Leigh Marshes matrix material and saltmarsh surface sediments in the Thames Estuary (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (mg $kg^{-1}$) (n=71) | Saltmarsh median (mg $kg^{-1}$) (n=91)[1] |
|---|---|---|---|---|
| Ag | 0.361 | No | 2 | 2 |
| Co | 0.069 | No | 38 | 35 |
| Cr | <0.001 | LM>sediment | 64 | 12 |
| Cu | <0.001 | LM>sediment | 605 | 31 |
| K | <0.001 | sediment>LM | 2177 | 4578 |
| Li | <0.001 | LM>sediment | 44 | 28 |
| Mg | <0.001 | sediment>LM | 2564 | 5717 |
| Mn | <0.001 | LM>sediment | 870 | 283 |
| Ni | <0.001 | LM>sediment | 141 | 35 |
| Pb | <0.001 | LM>sediment | 1332 | 61 |
| Sr | <0.001 | LM>sediment | 472 | 115 |
| Zn | <0.001 | LM>sediment | 1540 | 128 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data O'Shea, 2016, p. 232-233)

**Table 3.51: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between metal concentrations in the Hadleigh Marsh matrix material and saltmarsh surface sediments in the Thames Estuary (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Hadleigh Marsh median (mg $kg^{-1}$) (n=71) | Saltmarsh median (mg $kg^{-1}$) (n=91)[1] |
|---|---|---|---|---|
| Ag | <0.001 | sediment>HM | <LOD | 2 |
| Co | <0.001 | sediment>HM | 15 | 35 |
| Cr | <0.001 | HM>sediment | 59 | 12 |
| Cu | 0.111 | No | 33 | 31 |
| K | <0.001 | HM>sediment | 5713 | 4578 |
| Li | <0.001 | HM>sediment | 49 | 28 |
| Mg | <0.001 | HM>sediment | 7924 | 5717 |
| Mn | <0.001 | HM>sediment | 422 | 283 |
| Ni | <0.001 | HM>sediment | 45 | 35 |
| Pb | 0.340 | No | 63 | 61 |
| Sr | <0.001 | sediment>HM | 81 | 115 |
| Zn | <0.001 | HM>sediment | 209 | 128 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data O'Shea, 2016, p. 232-233)

### Inorganic contaminant concentrations in wood, paper and textiles compared to the matrix material, sediment quality guidelines and background concentrations

As materials eroded from landfills decompose they have the potential to become incorporated into bed sediments, where they can become a long-term source of contaminants to the aquatic environment and adversely affect bottom feeding and dwelling fauna (Canadian Council of Ministers of the Environment, 1995; Canadian Council of Ministers of the Environment, 2001). In marine waters it can be expected that wood will take a year or more to decompose, paper will decompose within 2-6 weeks, and textiles will take from 1 month (cottons) to hundreds of years (man-made fibres) (DHEC, nd).

Wood-boring and ingesting crustaceans and molluscs can be adversely affected by the presence of contaminants (Sleeter and Coull, 1973; Cragg et al., 1999; Pati et al., 2012). Paper fibres range from 300 to 500 µm in length and from 20 to 50 µm in diameter (Sun and Koziński, 2000) and therefore have the potential to be consumed by filter feeders, which are known to ingest particles up to 600 µm in diameter, and their contaminants

may be accumulated in the fauna (Yusof et al., 2004; Cefas, 2008). No specific standards exist to compare these effects against.

Contaminant concentrations in the wood, paper and textiles are compared with concentrations in the matrix material, NOAA guideline background sediment concentrations and Canadian sediment quality guidelines for Cd, Cr, Cu, Pb and Zn in Figure 3.21 (Buchman, 2008; Canadian Council of Ministers of the Environment, 2002). Table 3.52 shows the percentage of wood, paper and textiles samples from each site that exceed the ISQGs and PELs.

In the samples obtained for this research, wood contains significantly higher concentrations of Zn than the matrix material, comparable concentrations of Cu and Pb, and lower concentrations of the other metals (where there were sufficient data to determine rankings). The quantities of wood present in typical household waste are negligible (Figure 3.2) and no data is available relating to typical proportions of wood in other waste types, therefore it is not possible to assess the contribution of contaminants in wood to the total contaminant load for historic landfill sites. However, the wood samples contain Cu concentrations that exceed the ISQGs in over 50% of samples, Pb concentrations that exceed the PELs in over 50% of samples, and Zn concentrations that exceed the PELs in approximately 90% of samples.

The dry mass of paper in 1980s household waste, i.e. when Hadleigh Marsh landfill was in use, is typically greater than the dry mass of matrix materials by approximately 50% (Table 3.44). Based on typical proportions of paper in landfill sites of the era, it is likely that in the paper the total load of Cd is ~33%, Cr is ~24%, Cu is ~59%, Pb is ~45% and Zn is ~122% of the total load of those metals in the matrix material. Therefore, contaminants in paper are likely to make a significant contribution to the total contaminant load of the site and, consequently, provide a significant source of contamination if the waste is eroded and released into estuarine or coastal waters. The paper samples contain a significant proportion of samples with Cu and Pb concentrations that exceed the ISQGs, and Zn concentrations that exceed the PELs in about one-third of the samples.

The dry mass of textiles in 1980s household waste is typically about 17% of the dry mass of matrix materials. Based on typical proportions of textiles in landfill sites of the

era, it is likely that in the textiles the total load of Cd is ~6%, Cr is ~5%, Cu is ~52%, Pb is ~47% and Zn is ~42% of the total load of those metals in the matrix material. Therefore contaminants in textiles are also likely to make a significant contribution to the total contaminant load of the site and, consequently, provide a significant source of contamination if the waste is eroded and released into estuarine or coastal waters. About one-third of textile samples contain Cr concentrations that exceed the ISQGs, over 85% of textile samples exceed the ISQGs for Cu, and over 50% of textile samples contain Pb and Zn concentrations that exceed the PELs.

Sediments in the Thames Estuary are already polluted (Table 3.49) and contain significantly higher concentrations of most metals compared to the wood, paper and textile samples (Table 3.53, Table 3.54 and Table 3.55 respectively). Only Pb and Zn in the wood, and Cu, Pb and Zn in the textile samples are present at higher concentrations than in the saltmarsh sediments. This suggests that, although the matrix material is usually the focus of investigations into the potential environmental impacts of landfills, eroded wood, paper and textiles from landfills also have the potential to cause adverse ecological effects as they decompose and become incorporated into sediments, although in the Thames Estuary it is likely that of these only the wood and textiles would increase metal concentrations in the sediments above current levels. This highlights that the matrix material should not be the only material considered when determining the potential for pollution from historic coastal landfill sites.

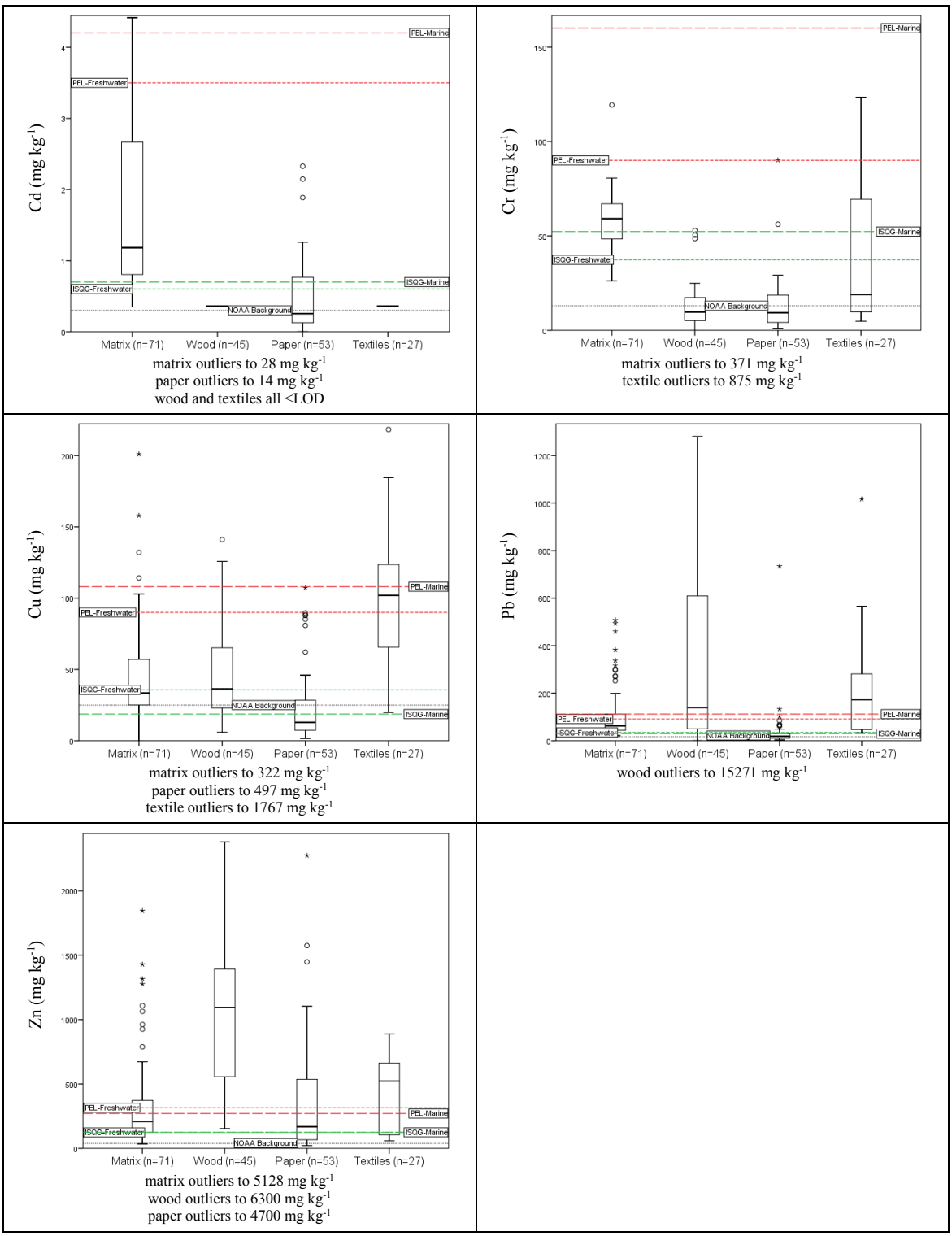



**Figure 3.21: Inorganic contaminant concentrations in wood, paper and textiles compared to concentrations in the matrix material and NOAA background levels for sediment and Canadian sediment quality guidelines (where available)**

**Table 3.52: Percentage of wood, paper and textile samples from Hadleigh Marsh landfill that exceed the Canadian sediment quality guidelines for metals**

| | Wood (n=45) | | | | Paper (n=53) | | | | Textiles (n=27) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Percentage of samples >ISQG | | Percentage of samples >PEL | | Percentage of samples >ISQG | | Percentage of samples >PEL | | Percentage of samples >ISQG | | Percentage of samples >PEL | |
| **Element** | **fresh** | **marine** | **fresh** | **marine** | **fresh** | **marine** | **fresh** | **marine** | **fresh** | **marine** | **fresh** | **marine** |
| Cd | Median<LOD | | 0 | 0 | 4 | 4 | 4 | 4 | Median<LOD | | 0 | 0 |
| Cr | 7 | 2 | 0 | 0 | 4 | 4 | 2 | 0 | 37 | 30 | 22 | 11 |
| Cu | 51 | 80 | 0 | 11 | 28 | 41 | 2 | 4 | 85 | 100 | 15 | 41 |
| Pb | 84 | 84 | 60 | 53 | 20 | 24 | 6 | 4 | 93 | 100 | 56 | 52 |
| Zn | 100 | 100 | 89 | 93 | 61 | 61 | 35 | 39 | 70 | 70 | 63 | 63 |

**Table 3.53: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between metal concentrations in the Hadleigh Marsh wood samples and saltmarsh surface sediments in the Thames Estuary (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Wood median (mg $kg^{-1}$) (n=71) | Saltmarsh median (mg $kg^{-1}$) (n=91)[1] |
|---|---|---|---|---|
| Ag | <0.001 | sediment>HM | <LOD | 2 |
| Co | <0.001 | sediment>HM | <LOD | 35 |
| Cr | <0.001 | sediment>HM | 10 | 12 |
| Cu | 0.162 | No | 36 | 31 |
| K | <0.001 | sediment>HM | 535 | 4578 |
| Li | <0.001 | sediment>HM | <LOD | 28 |
| Mg | <0.001 | sediment>HM | 703 | 5717 |
| Mn | <0.001 | sediment>HM | 84 | 283 |
| Ni | <0.001 | sediment>HM | 13 | 35 |
| Pb | 0.001 | HM>sediment | 140 | 61 |
| Sr | <0.001 | sediment>HM | 18 | 115 |
| Zn | <0.001 | HM>sediment | 1094 | 128 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data O'Shea, 2016, p. 232-233)

**Table 3.54: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between metal concentrations in the Hadleigh Marsh paper samples and saltmarsh surface sediments in the Thames Estuary (critical p value = 0.05)**

| Element | Mann-Whitney U p value | Significant difference | Paper median (mg $kg^{-1}$) (n=71) | Saltmarsh median (mg $kg^{-1}$) (n=91)[1] |
|---|---|---|---|---|
| Ag | <0.001 | sediment>HM | 0.1 | 2 |
| Co | <0.001 | sediment>HM | 1 | 35 |
| Cr | <0.001 | sediment>HM | 9 | 12 |
| Cu | <0.001 | sediment>HM | 13 | 31 |
| K | <0.001 | sediment>HM | 414 | 4578 |
| Li | <0.001 | sediment>HM | 3 | 28 |
| Mg | <0.001 | sediment>HM | 702 | 5717 |
| Mn | <0.001 | sediment>HM | 22 | 283 |
| Ni | <0.001 | sediment>HM | 5 | 35 |
| Pb | <0.001 | sediment>HM | 19 | 61 |
| Sr | <0.001 | sediment>HM | 22 | 115 |
| Zn | 0.058 | No | 168 | 128 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data O'Shea, 2016, p. 232-233)

**Table 3.55: Results of the Mann-Whitney U tests applied to determine whether there were significant differences between metal concentrations in the Hadleigh Marsh textile samples and saltmarsh surface sediments in the Thames Estuary (critical p value = 0.05)**

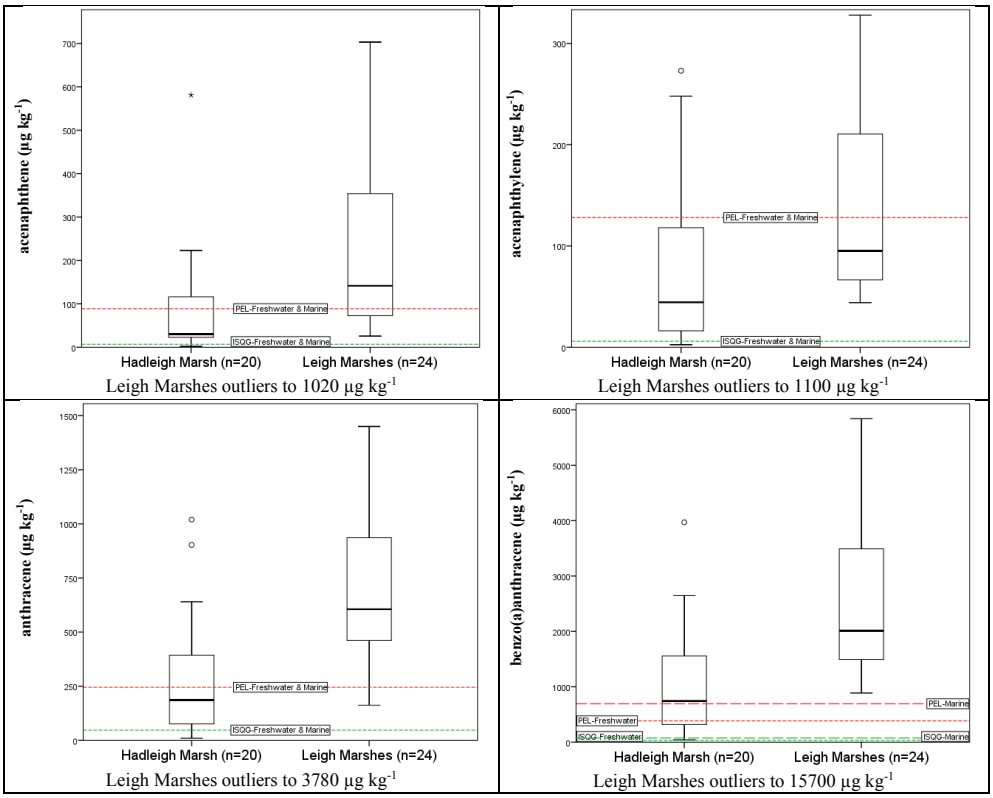
| Element | Mann-Whitney U p value | Significant difference | Textiles median (mg $kg^{-1}$) (n=71) | Saltmarsh median (mg $kg^{-1}$) (n=91)[1] |
|---|---|---|---|---|
| Ag | <0.001 | sediment>HM | 0.5 | 2 |
| Co | <0.001 | sediment>HM | <LOD | 35 |
| Cr | 0.548 | No | 19 | 12 |
| Cu | <0.001 | HM>sediment | 102 | 31 |
| K | <0.001 | sediment>HM | 676 | 4578 |
| Li | <0.001 | sediment>HM | <LOD | 28 |
| Mg | <0.001 | sediment>HM | 1100 | 5717 |
| Mn | 0.611 | No | 273 | 283 |
| Ni | <0.001 | sediment>HM | 17 | 35 |
| Pb | 0.007 | HM>sediment | 174 | 61 |
| Sr | 0.463 | No | 123 | 115 |
| Zn | 0.003 | HM>sediment | 522 | 128 |

[1]Surface sediments from a saltmarsh which is within 1.5 km of Leigh Marshes and Hadleigh Marsh landfill sites (raw data O'Shea, 2016, p. 232-233)

**Organic contaminant concentrations in the matrix material compared to sediment quality guidelines and background concentrations**

PAH concentrations in the matrix material are compared with Canadian sediment quality guidelines for acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(ah)anthracene, fluoranthene, fluorene, naphthalene, phenanthrene and pyrene in Figure 3.22 (Buchman, 2008; Canadian Council of Ministers of the Environment, 2002). NB No background sediment concentration data are published by NOAA for PAHs. Table 3.56 shows the percentage of samples from each site that exceed the ISQGs and PELs.

The majority of the measured contaminant concentrations exceed the ISQGs, and the majority of the Leigh Marshes samples and many of the Hadleigh Marsh samples exceed the PELs. Upstream of Leigh Marshes and Hadleigh Marsh surface sediments are already contaminated with PAHs (Table 3.57), but PAH concentrations in the Leigh Marshes and Hadleigh Marsh matrix material are higher, and downstream there is little PAH contamination of sediments (Woodhead et al., 1999; Chesman et al., 2006). This suggests that PAH concentrations in the eroded matrix material could have both an immediate adverse ecological impact locally and a long-term adverse ecological impact wherever the waste is dispersed (subject to any dilution effects).



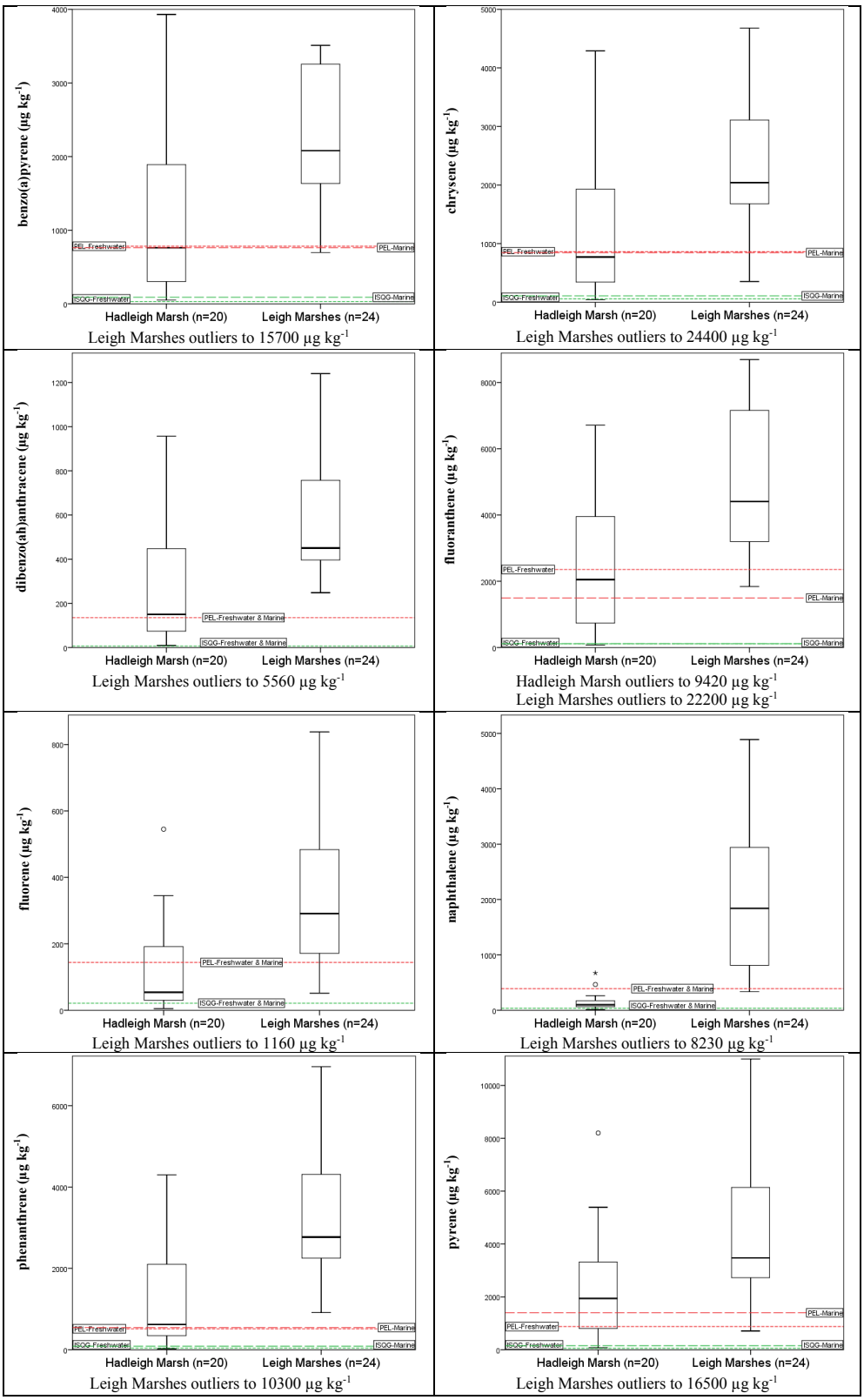



**Figure 3.22: Organic contaminant concentrations in the matrix material compared to Canadian sediment quality guidelines (where available)**

**Table 3.56: Percentage of matrix material samples that exceed the Canadian sediment quality guidelines for PAHs in Leigh Marshes and Hadleigh Marsh landfills**

| | Leigh Marshes (n=24) | | | | Hadleigh Marsh (n=20) | | | |
|---|---|---|---|---|---|---|---|---|
| | Percentage of samples >ISQG | | Percentage of samples >PEL | | Percentage of samples >ISQG | | Percentage of samples >PEL | |
| | fresh | marine | fresh | marine | fresh | marine | fresh | marine |
| acenaphthene | 100 | 100 | 58 | 58 | 90 | 90 | 35 | 35 |
| acenaphthylene | 100 | 100 | 38 | 38 | 95 | 95 | 25 | 25 |
| anthracene | 100 | 100 | 92 | 92 | 80 | 80 | 35 | 35 |
| benzo(a)anthracene | 100 | 100 | 100 | 100 | 100 | 95 | 70 | 50 |
| benzo(a)pyrene | 100 | 100 | 96 | 96 | 100 | 95 | 50 | 50 |
| chrysene | 100 | 100 | 96 | 96 | 95 | 95 | 45 | 45 |
| dibenzo(ah)anthracene | 100 | 100 | 100 | 100 | 100 | 100 | 55 | 55 |
| fluoranthene | 100 | 100 | 88 | 100 | 95 | 95 | 45 | 55 |
| fluorene | 100 | 100 | 79 | 79 | 85 | 85 | 30 | 30 |
| naphthalene | 100 | 100 | 96 | 96 | 85 | 85 | 10 | 10 |
| phenanthrene | 100 | 100 | 100 | 100 | 95 | 90 | 50 | 50 |
| pyrene | 100 | 100 | 96 | 96 | 100 | 95 | 65 | 55 |
| NB Canadian guidelines do not specify Interim Sediment Quality Guidelines (ISQGs) or Probable Effect Levels (PELs) for the other PAHs measured. | | | | | | | | |

**Table 3.57: Organic contaminant concentrations in Leigh Marshes (LM) and Hadleigh Marsh (HM) matrix material compared to surface sediments in the Thames Estuary (light grey > marine ISQG, dark grey > marine PEL, NB values that exceed PELs also exceed ISQGs)**

| | Matrix material | | Surface sediments in the Thames Estuary[1] | | | | |
|---|---|---|---|---|---|---|---|
| | | | Upstream of LM and HM | | Downstream of LM and HM | | |
| PAH (µg kg$^{-1}$) | LM median | HM median | West Thurrock 1996 | Mucking 1996 | Warp 1993 | Warp 1996 | Outer Gabbard 1996 |
| acenaphthene | 142 | 30 | <4 | 21 | <4 | <4 | <4 |
| acenaphthylene | 95 | 44 | | | | | |
| anthanthrene | 479 | 156 | | | | | |
| anthracene | 606 | 186 | 79 | 64 | 11 | 15 | <1 |
| benzo(b+k)fluoranthene | 3755 | 1435 | | | | | |
| benzo(a)anthracene | 2010 | 743 | 283 | 227 | 68 | 51 | <2 |
| benzo(a)pyrene | 2080 | 761 | 717 | 395 | 123 | 60 | <2 |
| benzo(b)fluoranthene | 2910 | 1055 | 750 | 452 | 88 | 56 | <3 |
| benzo(e)pyrene | 1575 | 567 | 1070 | 777 | <3 | 98 | <3 |
| benzo(ghi)perylene | 1570 | 577 | 777 | 419 | 94 | 45 | <5 |
| benzo(k)fluoranthene | 930 | 407 | 363 | 204 | 80 | 24 | <1 |
| chrysene | 2040 | 771 | 259 | 196 | 72 | 44 | <2 |
| coronene | 386 | 144 | | | | | |
| cyclopenta(cd)pyrene | <10 | <10 | | | | | |
| dibenzo(ah)anthracene | 451 | 150 | 58 | 69 | <3 | 39 | <3 |
| fluoranthene | 4405 | 2050 | 670 | 485 | 83 | 96 | <4 |
| fluorene | 291 | 54 | <15 | <23 | <15 | <15 | <15 |
| indeno(1,2,3-cd)pyrene | 1885 | 853 | | | | | |
| naphthalene | 1840 | 100 | 116 | 80 | <13 | 13 | <13 |
| perylene | 466 | 211 | | | | | |
| phenanthrene | 2770 | 622 | 482 | 253 | 83 | 60 | <11 |
| pyrene | 3475 | 1940 | 895 | 482 | 69 | 114 | <2 |

[1]PAH concentrations in Thames Estuary surface sediments (Woodhead et al., 1999 p. 776-777)

## *Summary*

This research has found that inorganic and organic contaminant concentrations in materials in Hadleigh Marsh and Leigh Marshes landfills are highly variable. Differences of up to four orders of magnitude between minimum and maximum contaminant concentrations were found within individual trial pits, and significant differences in contaminant concentrations between different areas of the same site and between sites were also found. The results are comparable to the limited existing data, but this reflects the wide range of results in previous studies rather than any consistency in contaminant concentrations between sites. These highly variable contaminant concentrations are probably due to the highly heterogeneous nature of the landfilled waste. Sources of contaminants are likely to be numerous and there is evidence that metals in the waste have been redistributed by leachate movement. This heterogeneity means that contaminant datasets from one landfill site should not be assumed to be representative of other similar sites or representative of landfill sites in general. This has implications for undertaking landfill risk assessments as sampling and analyses are resource intensive tasks which some landfill risk assessment methods aim to avoid by using a single generic contaminant dataset for screening level risk assessments (e.g. Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013). The findings of this research suggest that risk screening assessments should consider the relative risks of different types of waste present within the sites, i.e. household, commercial, industrial, etc., (after Alaska Department of Environmental Conservation, 2015) rather than assuming all sites contain the same contaminant concentrations and materials. The relative risk of different types of waste is considered as part of the development of a risk screening assessment method in Chapter 5.

Generally, contaminant concentrations exceed sediment quality guidelines and may pose a significant threat to the environment if the sites are not adequately maintained and waste materials are allowed to erode. Soil and soil-like materials are the usual focus of contaminated land investigations, but contaminant concentrations in wood, paper and textiles are also potentially hazardous and may be sources and sinks for contaminants in the waste stream. This highlights that contaminated land investigations and landfill risk assessments should not just focus on soil and soil-like materials, but should also

consider other materials that are present. The importance of this will vary depending on the mixture of materials present in individual landfill sites.

Accurately determining the environmental impact resulting from eroded waste requires knowledge of the rate of waste erosion and dispersion, but currently there are no suitable tools for modelling landfill erosion and understanding of dispersion is limited (Browne et al., 2015). However, the landfills are within or adjacent to designated sites where it is an offence to allow any pollution to occur (see section 3.2.3). The response strategy that has been used at landfill sites that have already breached and released waste, e.g. at Lyme Regis, UK, is to collect the largest waste materials as they erode. This has left behind contaminated materials that have resulted in local increases in metal concentrations in sediments and biota (Pope et al., 2011). If the same response strategy were used for Leigh Marshes or Hadleigh Marsh in the event of a breach it would leave behind highly contaminated matrix materials which would breach legislation in place to protect the designated sites. It would also be difficult or impossible to implement such a strategy at sites with narrow foreshores before materials start to be transported away from the sites. Similarly, at least 443 other historic coastal landfills fall in or within 100 m of designated ecological sites where it is an offence to allow any pollution to occur.

Therefore, it is clear that in the short-term these landfill sites and their defences must be maintained to prevent any contaminated materials being released. In the long-term it may be necessary to relocate the waste if climate change pressures prohibit continued maintenance – a strategy the US Government is already applying in Alaska (State of Alaska Department of Environmental Conservation, 2012). Removing waste from the sites is prohibitively expensive, the landfill tax alone to relocate Hadleigh Marsh would be circa £23-30 million (landfill tax = £84.40 per tonne at the time of writing, HM Revenue and Customs, 2016). Before such expenditures can be justified, further research is required into alternative long-term management options, and into the mechanical stability and life expectancy of historic coastal landfill sites, in order to identify criteria that can determine when and if the management approach should switch from site maintenance to removal of the waste. It is also necessary to understand which landfill sites pose the greatest pollution risk in order that management resources can be prioritised, a risk screening assessment method is proposed in Chapter 5.

# Chapter 4. Contamination of surface waters by soluble metals leaching from historic coastal landfill sites

## *Introduction*

## *4.1 Aims and objectives*

Metal concentrations in leachates generated by landfills through freshwater ingress under normal operating conditions, i.e. waste is fully contained by capping materials and not subject to inundation, are well studied (e.g. Robinson and Maris, 1979; Robinson et al., 1982; LaGrega et al., 1994; Robinson, 1995; 2007; Ziyang et al., 2009). However, little research exists relating to the mobilisation of metals from waste eroded into surface waters (Neuhold, 2013) or the effects on metal mobility of the ingress of estuarine or marine waters into landfill waste (Khoury et al., 2000). Therefore, it is unknown what effect inundation of historic coastal landfills by estuarine or marine waters would have on metal concentrations in leachates. It is also unknown whether metals could be released to the water column at concentrations that would cause pollution in the event of a historic coastal landfill being eroded or catastrophically breached.

In addition, existing landfill risk assessments that attempt to rate the pollution risk from the release of metals from waste into surface waters have assumed that all landfill sites release the same proportion of their metal content into solution (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013). Therefore, understanding whether there are significant differences in metal mobilisation between landfill sites is necessary to determine whether this approach is appropriate when assessing the pollution risk.

The aim of this chapter is to determine the potential for contamination of surface waters by metals released from landfill waste that is either inundated by flood water or eroded into surface waters. This has been investigated by addressing the following objectives, to:

1. Determine whether there are differences in the proportions of metals released into water from waste samples from two different historic landfill sites.

2. Determine whether there are differences in the proportions of metals released from waste into freshwater and seawater.
3. Determine the mass of metals released per kilogram of waste to enable comparison to Waste Acceptance Criteria limit values as a measure of the impact upon water quality from inundated, contained waste.
4. Determine the potential concentrations of metals in waters that landfill waste has eroded into, which will enable comparison to Environmental Quality Standards for the Protection of Surface Water Quality as a measure of the impact upon water quality.

## 4.2 Overview of methodological approaches used

To determine metal concentrations released to solution if the landfill sites are subject to fluvial or coastal inundation, or if waste is eroded and released into fluvial or coastal waters, leaching experiments were carried out using matrix material from waste excavated from Hadleigh Marsh and Leigh Marshes landfills. The resulting leachates were then analysed to determine metal concentrations, and the mass of metal released per kilogram of matrix material was then calculated to enable comparison to Waste Acceptance Criteria limit values.

To calculate the proportion of the initial metal concentration in the matrix material that was released during leaching, it was necessary to determine the metal concentrations in the matrix material before and after leaching. Acid extractions on additional subsamples are commonly used to approximate initial (pre-leaching) metal concentrations (e.g. Acosta et al., 2011; Wang et al., 2015b). However, to minimise errors relating to the highly variable metal concentrations in the matrix material (see Chapter 3), the initial metal concentrations in the matrix material were indirectly determined by mass-balance calculations using metal concentrations measured in the leachate and the post-leaching matrix material. To further reduce heterogeneity related errors, metal concentrations in the post-leaching matrix material were determined on triplicate subsamples of each sample leached.

Figure 4.1 shows a process map that summarises the methods used. Sections 4.3 to 4.13 detail the methods used, the reasons for their selection and the necessary calculations.

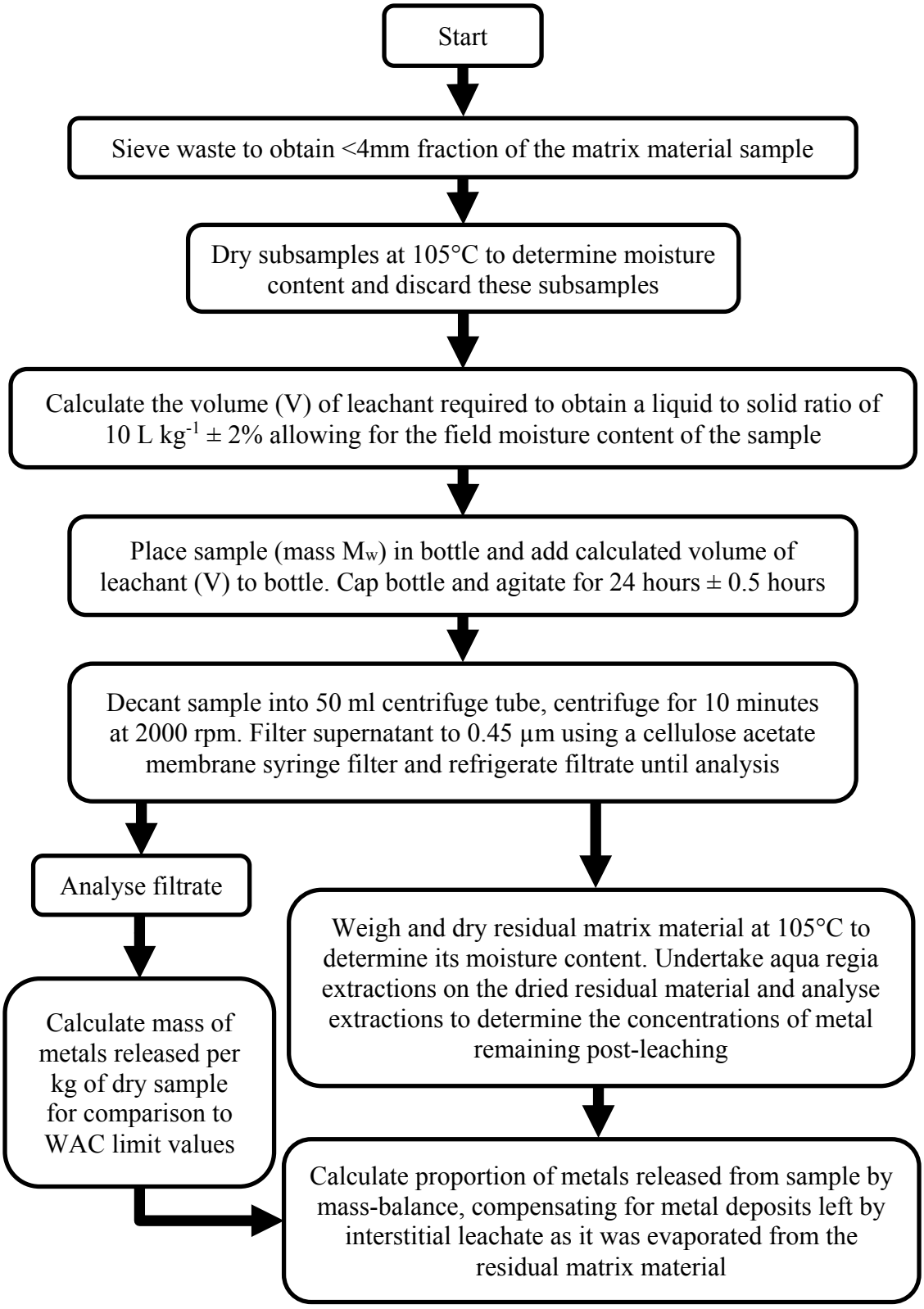



**Figure 4.1: Leaching method process map**

## 4.3 Selection of a leaching methodology

Typically Environmental Quality Standards (EQSs) and associated discharge permitting standards are used by researchers to determine whether metal concentrations in leachates are likely to adversely affect water quality (e.g. Kumar and Alappat, 2005; Robinson, 2007). However, to compare metal concentrations in leachates to EQSs it is necessary to know the ratios of water to landfill waste that accurately represent real-world events and accurate determinations of these ratios are difficult to achieve when considering infiltration of water into a landfill site. Hence, any direct comparison of concentrations of metals in laboratory generated leachates to EQSs would be meaningless. In the landfill industry, samples of waste being sent to present-day landfill sites are tested using standardised leaching ratios and methods defined in BS EN 12457-1, -2 and -4 (British Standards Institution (BSI), 2002a; 2002b; 2002c) to determine their potential to adversely affect water quality if leachates leak from the sites. The masses of metals released per kilogram of waste during the tests are compared to Waste Acceptance Criteria (WAC) limit values to categorise the waste as inert, non-hazardous or hazardous (Council Decision, 2003; Council Directive, 1999). As leachate from inert waste is considered unlikely to have any adverse impact upon the water environment BS EN 12457-1, -2 and -4 provide a means of assessing whether waste would release ecologically harmful levels of metals without having to quantify a real-world leaching ratio. The use of a commercially available, standard industry method also has the benefit of allowing landfill managers to easily replicate this methodology when assessing the pollution risk posed by other historic coastal landfill sites. Additionally, assessing the metals against the WAC will provide landfill managers with details of which category of present-day landfill the waste would be accepted at if it needed to be moved to a less vulnerable site, and also enables determination of whether the landfill has reached Final Storage Quality (FSQ) against the proposed criteria that FSQ waste should be similar to inert waste (Scharff et al., 2007; Valencia et al., 2009).

The WAC limit values and leaching methods were developed for the protection of groundwater, but are also used for assessing risk to surface waters (Defra, 2010). Deionised water is commonly used as a proxy for river water (Millward and Liu, 2003; Suh et al., 2003; Wong et al., 2010; 2013) and BS EN 12457-1, -2 and -4 specify that the leachant (i.e. water used for leaching) should be distilled, demineralised or deionised

water or water of equivalent purity (5<pH<7.5) with a conductivity <0.5 mSlm as a proxy for groundwater and surface water. However, historic landfills are present in fluvial, estuarine and fully coastal environments and, therefore, this research used deionised water and artificial seawater as leachants to represent the extremes of the salinity gradient along the catchment to coast continuum. Laboratory grade sea salts in deionised water (Table 4.1) were used to make artificial seawater (Wong et al., 2013). Buffers are sometimes added to leachants in order to maintain their pH during tests to represent the acidity at different stages of landfill stabilisation (e.g. Suna Erses and Onay, 2003), but BS EN 12457-1, -2 and -4 specify that no buffers should be added to the leachants and, therefore, no buffers were used in this research.

**Table 4.1: Composition of artificial seawater made by adding 35.14g of Sigma-Aldrich sea salts (product no. S9883) to 1 litre of deionised water**

| Components | Concentration (mg $L^{-1}$) | Components | Concentration (mg $L^{-1}$) |
|---|---|---|---|
| Chloride | 19,290 | Boron | 5.6 |
| Sodium | 10,780 | Bromide | 56 |
| Sulphate | 2,660 | Iodide | 0.24 |
| Potassium | 420 | Lithium | 0.3 |
| Calcium | 400 | Fluoride | 1.0 |
| Magnesium | 1,320 | Carbonate (Bicarbonate) | 200 |
| Strontium | 8.8 | Other trace elements | < 0.5 |

All three BS leaching methods specify that the leaches are carried out for 24 hours. BS EN 12457-1 specifies the use of the <4 mm fraction with a leaching ratio of 2 litres of leachant per kg of dry waste, BS EN 12457-2 specifies the use of the <4 mm fraction with a leaching ratio of 10 litres of leachant per kg of dry waste, and BS EN 12457-4 specifies the use of the <10 mm fraction with a leaching ratio of 10 litres of leachant per kg of dry waste respectively. Little information is given as to why there are different methods and the choice of which to use is left to the discretion of the analyst. The two leaching ratios were trialled and it was found that the 10 litres per kg of waste leaching ratio consistently resulted in greater metal release per kg of waste than the 2 litres per kg leaching ratio (see Figure 4.2). Two possible reasons for this are: first, reducing the leachant to waste ratio may have limited metal release due to concentrations of the metals reaching saturation in the leachate (Fällman and Aurell, 1996); second, it is likely there was resorption of metals during filtering of the 2 litres per kg leachate as at

that ratio the leachate and matrix material formed a sludge and it was not possible to decant the leachate into the filter without also decanting the majority of the matrix material. The 10 litres per kg leachate rapidly separated into matrix and leachant layers once agitation was stopped, which allowed just the leachate to be poured onto the filter reducing the opportunity for resorption. Therefore, 10 litres of leachant per kg of dry waste was selected as the preferred leaching ratio as it generates the worst case concentrations of metals in leachates. The 10 litres per kg of material leaching ratio has been commonly used in studies of sediment (e.g. Ham et al., 1979; Belevi and Baccini, 1989; Wong et al., 2010) as it represents materials with a high throughput of water such as waste being released into the intertidal zone through site erosion (Wahlström, 1996). The <4 mm fraction was selected in preference to the <10 mm fraction as it is more readily eroded and transported (Mayhew, 2015) and, hence, better represents the material likely to be released from an eroding landfill. Therefore, BS EN 12457-2 was selected as the leaching method as it specifies the use of the <4 mm fraction with a leaching ratio of 10 litres of leachant per kg of dry waste.

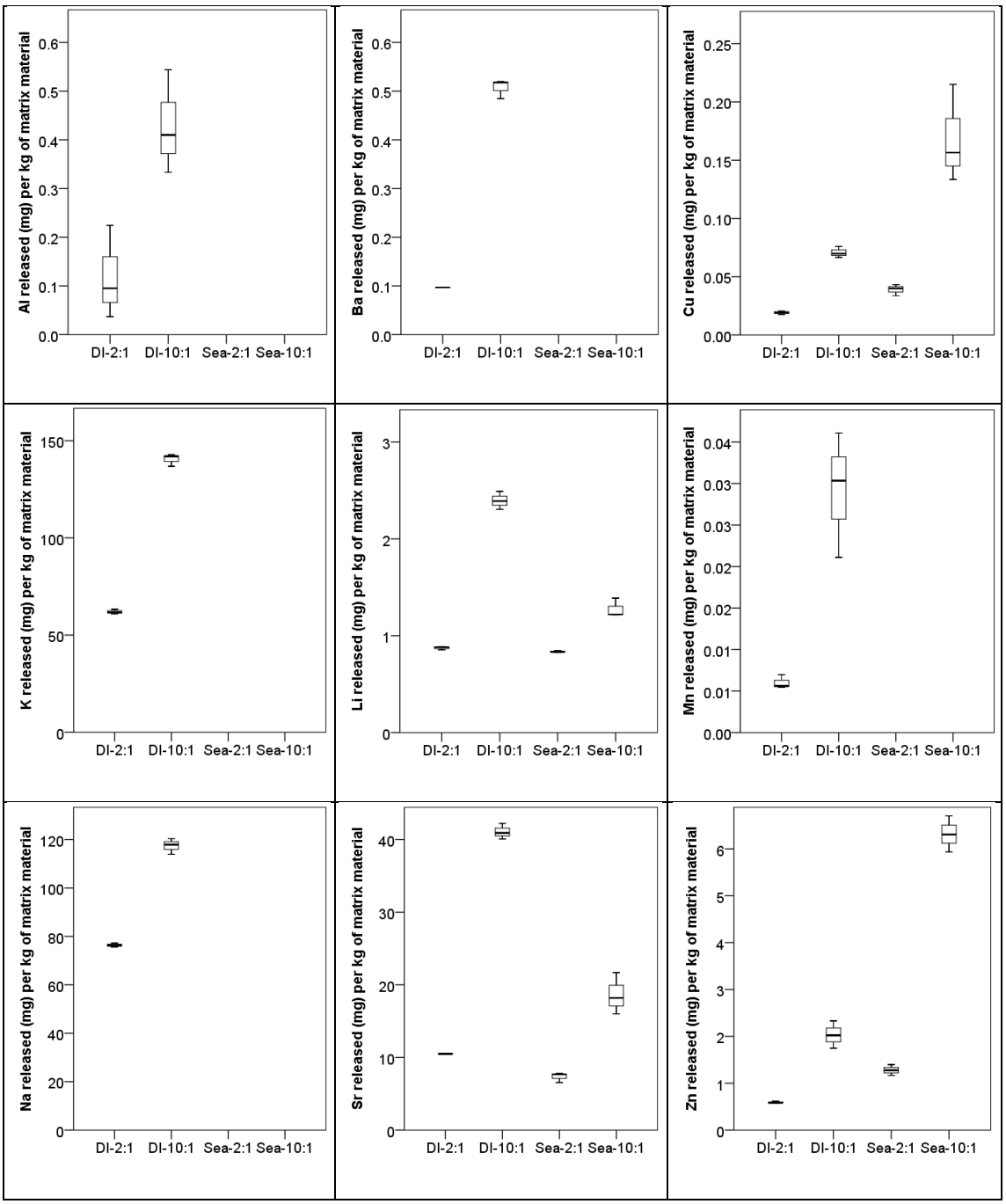



**Figure 4.2: Comparison of the metals released per kilogram of waste at leaching ratios of 2 litres and 10 litres of leachant per kg of dry waste. NB Al, Ba, K, Mn and Na were below detection limits in the seawater leaches. DI=Deionised water leaches. Sea=Artificial seawater leaches. 2:1 and 10:1 = liquid (L) to solid (kg) leaching ratios**

## 4.4 Selection of material type to be leached

Waste extracted from Leigh Marshes and Hadleigh Marsh landfills consisted of a mixture of materials (see Chapter 3). BS EN 12457-2 requires the leaching of a single, mixed sample of each batch of waste being investigated. However, leaching tests are sensitive to inhomogeneities in the materials being tested (Fällman and Aurell, 1996) and metal concentrations in the different materials in the landfill waste are highly heterogeneous (see Chapter 3). Therefore, to eliminate variation in results that could be caused by differences in sample mixtures rather than the leachant type, a single material type was selected for the leaching experiments. The matrix material was selected as it is the most abundant material present (by mass) in typical household waste since the 1930s (Bridgwater, 1986; Parfitt, 2009) and it carries the majority of the metal load (see Chapter 3).

Homogenised matrix material samples for Hadleigh Marsh trial pit 3 and for Leigh Marshes trial pit 5 were created. These trial pits were selected randomly, one from each site. Waste from each inorganic sample container for trial pit 3 and trial pit 5 was sieved into plastic bags until 100 g of the <4 mm fraction had been separated from each container. Material over 4 mm was returned to its sample container. Between samples the sieve was washed, rinsed with deionised water and dried to prevent cross-contamination of samples. One large plastic bag per trial pit was then used to mix the samples, creating a 400 g homogenised sample to represent each site. To retain field moisture the bags were then sealed until the samples were required for leaching.

## 4.5 Moisture content

### 4.5.1 Pre-leaching moisture content

BS EN 12457-2 requires the use of field moist samples with the addition of a known volume of leachant to achieve a leaching ratio of 10 litres of leachant (composed of the added leachant and the sample's field moisture) per kg of dry waste (see section 4.6). Therefore, subsamples of the homogenised matrix material were dried to enable calculation of the field moisture content on a dry mass basis.

For each of the homogenised samples, three subsamples of 10 g of matrix material, sieved to <4 mm, were weighed into pre-weighed crucibles and air dried overnight and then on a hotplate for 6 hours at 105 °C in a fume cupboard. Trial runs indicated no further decrease in mass after 6 hours of drying when measured to an accuracy of 10 mg. The dried samples were then reweighed and the initial moisture content was calculated using Equation 3.1, the results can be found in Appendix B.

**Equation 4.1: Calculation of moisture content in waste materials on a dry mass basis (ISO, 1993)**

$$Moisture\ content\ (\%) = \left(\frac{wet\ mass\ of\ material - dry\ mass\ of\ material}{dry\ mass\ of\ material}\right)\ x\ 100\%$$

### 4.5.2 Post-leaching moisture content

The moisture content in the post-leaching matrix material was determined in order that the concentration of metals left as residue by the evaporated interstitial leachate could be calculated. This allowed the origin of metals present in the post-leaching material to be apportioned between the interstitial leachate and the solid material, which is necessary to accurately calculate the concentrations of metals present in the material prior to leaching (section 4.9). To determine the moisture content post-leaching, the matrix material was transferred to pre-weighed, acid washed crucibles using disposable plastic spatulas, weighed, dried (see section 4.5.1), reweighed, and then stored in sealed plastic bags until later analysis of metal concentrations.

## 4.6 Leaching

The proportion of metals released from the same material under the same conditions can vary significantly (Svensson et al., 2005); therefore, replicate samples of each of the homogenized materials were leached. All glassware and plasticware used for the leaches were acid washed (10% $HNO_3$), rinsed three times in deionised water and dried before use. 24 x 10 g (±0.5 g) homogenised matrix material samples from Leigh Marshes trial pit 5, and 24 x 10 g (±0.5 g) homogenised matrix material samples from Hadleigh Marsh trial pit 3 were leached in 100 ml plastic bottles, the samples were evenly divided into batch leach tests using deionised water and artificial seawater leachants. Drying materials can change metal mobility through oxidation (Tack et al., 1996; Hu et al., 2008); consequently, BS EN 12457-2 requires the use of field moist samples with the addition of a known volume of leachant to achieve a leaching ratio of 10 litres of leachant per kg of dry waste. The necessary volume of leachant to be added to 10 g of field moist matrix material was calculated using Equation 4.2 to be 64 ml for the Hadleigh Marsh samples and 55 ml for the Leigh Marshes samples.

**Equation 4.2: Calculation of the leachant volume (V) to be added to create a 10 litres of leachant to 1 kg of dry waste leaching ratio (after British Standards Institution (BSI), 2002b)**

$$V = \left(10 - \frac{MC}{100}\right) x \frac{M_w}{\left(1 + \frac{MC}{100}\right)}$$

*Where:*
*$V$ = Leachant volume (l)*
*$MC$ = Moisture content of material on a dry mass basis (Equation 3.1 in section 4.5)*
*$M_w$ = Wet mass (field moisture) of material to be leached (kg) (see section 4.5)*

A minimum of one blank sample (containing leachant, but no sample) per leachant type was included for each ten replicate samples leached to determine background contaminant levels. Dissolved oxygen concentration (YSI 550A dissolved oxygen meter), temperature, redox potential and pH (VWR pH110 meter with pH/temp probe and ORP electrode) were measured in 25% of the leachates at the start and at the end of the leaching period prior to centrifuging (OECD, 2000; British Standards Institution (BSI), 2002b). The suspensions were agitated for 24 hours using an orbital shaker (VWR Standard Analogue Shaker 3500) (British Standards Institution (BSI), 2002b; Schäfer et al., 2009; Zhao et al., 2013). The agitation was such that the sample was kept in suspension for the duration of the leach. The samples were then decanted into 50 ml

centrifuge tubes and centrifuged for 10 minutes at 2000 rpm (Eppendorf centrifuge model 5804) and the supernatants filtered to 0.45 μm using cellulose acetate membrane syringe filters (Radojevic et al., 2006). The filtered leachates were then refrigerated until analysis.

## 4.7 Selection of metals for analysis

Waste Acceptance Criteria limit values exist for As, Ba, Cd, $Cr_{total}$, Cu, Hg, Mo, Ni, Pb, Sb, Se and Zn. It should be noted, however, that the WAC limits are designed to prevent freshwater pollution and do not consider estuarine or marine waters, having been derived by modelling typical landfill impacts on groundwater (Defra, 2010). To determine which of the WAC metals are of concern in estuarine and marine waters UK EQSs were reviewed and As, Cd, $Cr_{total}$, Cu, Hg, Ni, Pb and Zn were found to have limits set for marine waters (Environment Agency, 2011). Therefore, Cd, $Cr_{total}$, Cu, Ni, Pb and Zn were identified and analysed as the primary contaminants of interest. In addition, Al, Co, Fe and Mn were analysed. Arsenic and Hg were not measured in the leachates as they are volatile and are likely to have been lost from the waste during storage before leaching was carried out.

## 4.8 Metal analysis

To enable mass-balance calculations of the initial (pre-leaching) metal concentrations in the matrix material, triplicate subsamples of the dried post-leaching matrix material were extracted in aqua regia and analysed to determine post-leaching metal concentrations (see Chapter 3 for methods). Triplicates of each subsample were used due to the matrix material being found to be highly heterogeneous in the extractions previously carried out to determine contaminant variability (see Chapter 3).

All leachates and aqua regia extractions of the post-leaching matrix material were analysed by ICP-MS (Thermo Scientific X Series 2) by Plymouth University. Artificial seawater leachates were diluted by a factor of five, using 2% nitric acid, to reduce the total dissolved solids to within the ICP-MS analyser's limits. The deionised water leaches and post-leaching matrix material extractions were not diluted. The method and instrument precision were measured using replicate extractions of the CRMs (BCR-143R), the mean relative standard deviation (%RSD) was 4.7%. The mean recovery for the CRM extractions was 91.7% with a mean %RSD of 3.8%.

Standard scientific practice is to use blank corrections to compensate for the risk of over-reporting metal concentrations due to contaminated equipment or interferences occurring during ICP analysis. However, BS EN 12457-2, specifies that blank corrections should not be used when determining metal concentrations in the leachates and in this study blank corrections of leachates were not found to have any impact upon whether metal concentrations exceeded the WAC inert limit values.

## 4.9 Calculating initial concentrations of metals in the matrix material and the proportion of metals released

Initial metal concentrations in the dry sample prior to leaching ($M_{intial}$) were indirectly determined using Equation 4.3. The proportion of each metal released was then calculated as per Equation 4.6 as a percentage of the initial concentration of each metal present in the sample.

**Equation 4.3: Mass-balance calculation of the metal concentration in dry waste prior to leaching**

$$M_{initial} = M_{released} + M_{post} - M_{deposited}$$

*Where:*
*$M_{initial}$ = Concentration of metal in dry waste prior to leaching (mg kg$^{-1}$)*
*$M_{released}$ = Mass of metal released during leaching (mg) per kg of dry waste (Equation 4.7)*
*$M_{post}$ = Concentration of metal in dried post-leaching waste material (mg kg$^{-1}$)*
*$M_{deposited}$ = Concentration of metal residue from evaporated interstitial leachate in dried post-leaching waste material (mg kg$^{-1}$) (Equation 4.4)*

**Equation 4.4: Calculation of the concentration of residual metals from evaporated interstitial leachate in the dried post-leaching waste material ($M_{deposited}$)**

$$M_{deposited} = C_{leachate}\, x\, V_{evaporated}$$

*Where:*
*$C_{leachate}$ = Concentration of metal in leachate (mg L$^{-1}$)*
*$V_{evaporated}$ = Volume of interstitial leachate evaporated per kg (dry mass) of post-leaching waste material when drying (L kg$^{-1}$) (Equation 4.5)*

**Equation 4.5: Calculation of the volume of interstitial leachate evaporated per kg (dry mass) of post-leaching waste material ($V_{evaporated}$)**

$$V_{evaporated} = \left(\frac{moisture\ content}{100\ x\ density\ of\ water}\right)$$

*Where:*
*density of deionised water = 1 kg L$^{-1}$*
*density of 35 ppt seawater = 1.025 kg L$^{-1}$ (Georgia State University, nd)*
*moisture content = mass of water per kg of dry post leaching waste material (%) see Equation 3.1*

**Equation 4.6: Calculation of the proportion of metal released**

$$Proportion\ of\ metal\ released\ (\%) = 100\ x\ \frac{M_{released}}{M_{total}}$$

*Where:*
*$M_{initial}$ = Concentration of metal in dry waste prior to leaching (mg kg$^{-1}$) Equation 4.3*
*$M_{released}$ = Mass of metal released during leaching (mg) per kg of dry waste (Equation 4.7)*

## 4.10 Calculating the mass of metal released during leaching per kilogram of dry waste for comparison to WAC limit values

To enable comparison to Waste Acceptance Criteria (WAC) limit values, the Landfill Directive and BS EN 12457-2 require that the leaching results are reported as the mass of metal released per kilogram of dry waste, not, as is conventional, concentrations of metals in leachates. These values were calculated using Equation 4.7.

**Equation 4.7: Calculation of the mass of metal released per kilogram of dry waste (after British Standards Institution (BSI), 2002b)**

$$M_{released} = C_{leachate} x \left(\frac{V}{M_{dry}} + \frac{MC}{100}\right)$$

*Where:*
*$M_{released}$ = mass of metal released (mg) per kg of dry waste*
*V = leachant volume (Equation 4.2)*
*$M_{dry}$ = calculated dry mass of sample before leaching (kg) (Equation 4.8)*
*$C_{leachate}$ = concentration of metal in leachate (mg $L^{-1}$)*
*MC = Moisture content of material (Equation 3.1)*

**Equation 4.8: Calculation of the dry mass of sample before leaching ($M_{dry}$) (after British Standards Institution (BSI), 2002b)**

$$M_{dry} = \frac{M_w}{\left(1 + \frac{MC}{100}\right)}$$

*Where:*
*$M_w$ = Wet mass (field moisture) of the material before it was leached (kg)*
*MC = Moisture content of material (Equation 3.1)*

## 4.11 Calculating the potential concentrations of metals in the receiving waters for comparison to EQSs

To enable comparison to EQSs, the potential concentrations of metals in the receiving waters as a result of metal release from eroded waste must be calculated. The concentrations were calculated using Equation 4.9.

**Equation 4.9: Calculation of the concentration of metal in the receiving waters**

$$C_{RW} = \frac{M_{metal}\ x\ 10^9}{V_{dilution}}$$

*Where:*
*$C_{RW}$ = concentration of the metal in the receiving waters (ng $L^{-1}$)*
*$M_{metal}$ = total mass of soluble metal released into the receiving waters (kg) (see Equation 4.10)*
*$V_{dilution}$ = volume of receiving waters ($m^3$)*

**Equation 4.10: Calculation of the total mass of metal released to the estuary**

$$M_{metal} = \frac{M_{landfill}\ x\ M_{released}}{1000}$$

*Where:*
*$M_{metal}$ = total mass of soluble metal released into the receiving waters (kg)*
*$M_{landfill}$ = total dry mass of matrix material in the landfill (tonnes) (see Chapter 3)*
*$M_{released}$ = mass of metal released during leaching (mg) per kg of dry waste*

## 4.12 Dissolved Organic Carbon analysis

Dissolved Organic Carbon (DOC) in leachate can increase Cd, Ni and Zn mobility (Christensen et al., 1996) and affect metal toxicity (Umar et al., 2010). Therefore, 5 ml aliquots of each leachate was analysed to determine its DOC content using a Skalar Formacs Combustion TOC Analyzer calibrated using potassium biphthalate standards of 0.1, 0.5, 1, 5, 10, 25 and 50 $\mu L^{-1}$. A blank (deionised water) was analysed between each seawater leachate to ensure the analyser was fully rinsed between samples. Instrument precision was assessed using triplicates of two leachates on each analysis run and achieved a mean %RSD of 2.7%.

## 4.13 Statistical methods

Nonparametric statistical methods have been used, see Chapter 3. Full datasets can be found in Appendix B.

## 4.14 DO, redox potential, temperature and pH measurements

Concentrations of DO, redox potential, temperature and pH at the start and end of the leaching experiments are shown in Table 4.2 and Table 4.3 for the Leigh Marshes and Hadleigh Marsh leachates respectively.

No significant differences were found between DO, redox potential, temperature or pH when statistical comparisons (Mann-Whitney U tests) between the following groups of data were made:

1. The blank and sample leachates for each site at the start of the leaching experiments, see Table 4.4 for statistical data.
2. The blank and sample leachates for each site at the end of the leaching experiments, see Table 4.4 for statistical data.
3. The Leigh Marsh and Hadleigh Marsh sample leachates at the start of the leaching experiments, see Table 4.4 for statistical data.
4. The Leigh Marsh and Hadleigh Marsh sample leachates at the end of the leaching experiments, see Table 4.4 for statistical data.
5. The Leigh Marshes deionised water and artificial seawater leachates at the start of the leaching experiments, see Table 4.5 for statistical data.
6. The Leigh Marshes deionised water and artificial seawater leachates at the end of the leaching experiments, see Table 4.5 for statistical data.
7. The Hadleigh Marsh deionised water and artificial seawater leachates at the start of the leaching experiments, see Table 4.5 for statistical data.
8. The Hadleigh Marsh deionised water and artificial seawater leachates at the end of the leaching experiments, see Table 4.5 for statistical data.
9. The Leigh Marshes leachates at the start and end of the leaching experiments see Table 4.6 for statistical data.
10. The Hadleigh Marsh leachates at the start and end of the leaching experiments see Table 4.6 for statistical data.

**Table 4.2: DO, redox potential, temperature and pH in the Leigh Marshes leachates**

| Time | Leachant | Sample | | DO (%) | Redox (mV) | Temp. (°C) | pH |
|---|---|---|---|---|---|---|---|
| Start - 0 hrs | Deionised water | Blank (n=2) | Min. | 94.7 | 292 | 22.4 | 5.4 |
| | | | Max. | 102.1 | 358 | 22.6 | 6.2 |
| | | | Range | 7.4 | 66 | 0.2 | 0.8 |
| | | | Median | 98.4 | 325 | 22.5 | 5.8 |
| | | Matrix (n=3) | Min. | 96.2 | 301 | 20.9 | 7.3 |
| | | | Max. | 104.5 | 351 | 22.1 | 7.3 |
| | | | Range | 8.3 | 50 | 1.2 | 0 |
| | | | Median | 97.5 | 307 | 22.1 | 7.3 |
| | Artificial seawater | Blank (n=2) | Min. | 98.8 | 145 | 23.1 | 8.1 |
| | | | Max. | 99.8 | 153 | 23.2 | 8.3 |
| | | | Range | 1.0 | 8.0 | 0.1 | 0.2 |
| | | | Median | 99.3 | 149 | 23.2 | 8.2 |
| | | Matrix (n=3) | Min. | 100.1 | 229 | 22.3 | 7.3 |
| | | | Max. | 102.7 | 311 | 22.6 | 7.4 |
| | | | Range | 2.6 | 82.0 | 0.3 | 0.1 |
| | | | Median | 101.8 | 253 | 22.5 | 7.4 |
| End - 24 hrs | Deionised water | Blank (n=2) | Min. | 98.1 | 266 | 24.9 | 5.7 |
| | | | Max. | 98.5 | 283 | 24.9 | 6.1 |
| | | | Range | 0.4 | 17.0 | 0.0 | 0.4 |
| | | | Median | 98.3 | 275 | 24.9 | 5.9 |
| | | Matrix (n=3) | Min. | 95.8 | 302 | 24.4 | 7.2 |
| | | | Max. | 111.0 | 310 | 24.8 | 7.4 |
| | | | Range | 15.2 | 8.0 | 0.4 | 0.2 |
| | | | Median | 96.6 | 302 | 24.6 | 7.3 |
| | Artificial seawater | Blank (n=2) | Min. | 96.2 | 126 | 24.4 | 8.2 |
| | | | Max. | 96.4 | 185 | 24.6 | 8.2 |
| | | | Range | 0.2 | 59.0 | 0.2 | 0.0 |
| | | | Median | 96.3 | 156 | 24.5 | 8.2 |
| | | Matrix (n=3) | Min. | 88.2 | 182 | 23.7 | 7.1 |
| | | | Max. | 98.5 | 255 | 24.3 | 7.2 |
| | | | Range | 10.3 | 73.0 | 0.6 | 0.1 |
| | | | Median | 95.0 | 220 | 24.2 | 7.1 |

**Table 4.3: DO, redox potential, temperature and pH in the Hadleigh Marsh leachates**

| Time | Leachant | Sample | | DO (%) | Redox (mV) | Temp. (°C) | pH |
|---|---|---|---|---|---|---|---|
| Start - 0 hrs | Deionised water | Blank (n=2) | Min. | 94.7 | 197 | 21.7 | 5.5 |
| | | | Max. | 94.8 | 216 | 21.8 | 5.8 |
| | | | Range | 0.1 | 19.0 | 0.1 | 0.3 |
| | | | Median | 94.8 | 207 | 21.8 | 5.6 |
| | | Matrix (n=3) | Min. | 96.7 | 188 | 21.4 | 7.8 |
| | | | Max. | 103.8 | 222 | 21.7 | 8.1 |
| | | | Range | 7.1 | 34.0 | 0.3 | 0.3 |
| | | | Median | 100.8 | 198 | 21.6 | 7.9 |
| | Artificial seawater | Blank (n=2) | Min. | 87.3 | 95 | 21.8 | 8.2 |
| | | | Max. | 89.6 | 118 | 22.3 | 8.3 |
| | | | Range | 2.3 | 23.0 | 0.5 | 0.1 |
| | | | Median | 88.5 | 107 | 22.1 | 8.2 |
| | | Matrix (n=3) | Min. | 89.4 | 114 | 22.0 | 7.6 |
| | | | Max. | 97.0 | 125 | 22.3 | 7.8 |
| | | | Range | 7.6 | 11.0 | 0.3 | 0.2 |
| | | | Median | 96.6 | 120 | 22.1 | 7.6 |
| End - 24 hrs | Deionised water | Blank (n=2) | Min. | 98.3 | 184 | 23.0 | 7.2 |
| | | | Max. | 98.9 | 189 | 23.1 | 7.5 |
| | | | Range | 0.6 | 5.0 | 0.1 | 0.3 |
| | | | Median | 98.6 | 187 | 23.1 | 7.3 |
| | | Matrix (n=3) | Min. | 101.8 | 151 | 22.7 | 7.3 |
| | | | Max. | 106.7 | 166 | 22.9 | 7.6 |
| | | | Range | 4.9 | 15.0 | 0.2 | 0.3 |
| | | | Median | 102.1 | 154 | 22.8 | 7.5 |
| | Artificial seawater | Blank (n=2) | Min. | 92.3 | 123 | 24.2 | 8.0 |
| | | | Max. | 98.5 | 127 | 24.2 | 8.1 |
| | | | Range | 6.2 | 4.0 | 0.0 | 0.1 |
| | | | Median | 95.4 | 125 | 24.2 | 8.0 |
| | | Matrix (n=3) | Min. | 95.8 | 146 | 23.7 | 7.1 |
| | | | Max. | 104.3 | 162 | 24.1 | 7.3 |
| | | | Range | 8.5 | 16.0 | 0.4 | 0.2 |
| | | | Median | 100.9 | 158 | 24.0 | 7.2 |

**Table 4.4: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in DO, redox potential, temperature and pH between the blanks and samples from Leigh Marshes and Hadleigh Marsh, and between the samples from the two sites (see tables above for medians and sample numbers)**

| Leachant type & comparison | Time (hrs) | p values (critical p value = 0.050) | | | |
|---|---|---|---|---|---|
| | | DO | Redox | Temp. | pH |
| Deionised water Leigh Marshes blanks to samples | 0 | 0.800 | 1.000 | 0.200 | 0.200 |
| | 24 | 0.800 | 0.200 | 0.200 | 0.200 |
| Artificial seawater Leigh Marshes blanks to samples | 0 | 0.200 | 0.200 | 0.200 | 0.200 |
| | 24 | 0.800 | 0.400 | 0.200 | 0.200 |
| Deionised water Hadleigh Marsh blanks to samples | 0 | 0.200 | 1.000 | 0.200 | 0.200 |
| | 24 | 0.200 | 0.200 | 0.200 | 0.400 |
| Artificial seawater Hadleigh Marsh blanks to samples | 0 | 0.400 | 0.400 | 0.800 | 0.200 |
| | 24 | 0.400 | 0.200 | 0.200 | 0.200 |
| Deionised water Leigh Marshes to Hadleigh Marsh samples | 0 | 1.000 | 0.100 | 0.700 | 0.100 |
| | 24 | 0.700 | 0.100 | 0.100 | 0.400 |
| Artificial seawater Leigh Marshes to Hadleigh Marsh samples | 0 | 0.100 | 0.100 | 0.100 | 0.100 |
| | 24 | 0.200 | 0.100 | 0.400 | 0.400 |

**Table 4.5: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in DO, redox potential, temperature and pH between the deionised water and artificial seawater leachates (see tables above for medians and sample numbers)**

| Site & comparison type | Time (hrs) | p values (critical p value = 0.050) | | | |
|---|---|---|---|---|---|
| | | DO | Redox | Temp. | pH |
| Leigh Marshes blanks Deionised water to artificial seawater | 0 | 1.000 | 0.333 | 0.333 | 0.333 |
| | 24 | 0.333 | 0.333 | 0.333 | 0.333 |
| Leigh Marshes samples Deionised water to artificial seawater | 0 | 0.700 | 0.400 | 0.100 | 0.100 |
| | 24 | 0.400 | 0.100 | 0.100 | 0.200 |
| Hadleigh Marsh blanks Deionised water to artificial seawater | 0 | 0.333 | 0.333 | 0.333 | 0.333 |
| | 24 | 0.667 | 0.333 | 0.333 | 0.333 |
| Hadleigh Marsh samples Deionised water to artificial seawater | 0 | 0.200 | 0.100 | 0.100 | 0.200 |
| | 24 | 0.400 | 0.100 | 0.100 | 0.200 |

**Table 4.6: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in DO, redox potential, temperature and pH between the leachates at the start and end of the leaching experiments (see tables above for medians and sample numbers)**

| Site, leachant type & comparison type | Material | p values (critical p value = 0.050) | | | |
|---|---|---|---|---|---|
| | | DO | Redox | Temp. | pH |
| Leigh Marshes deionised water 0 hours to 24 hours | Blanks | 1.000 | 0.333 | 0.333 | 1.000 |
| | Samples | 1.000 | 1.000 | 1.000 | 1.000 |
| Leigh Marshes artificial seawater 0 hours to 24 hours | Blanks | 0.333 | 1.000 | 0.333 | 1.000 |
| | Samples | 0.100 | 0.400 | 0.100 | 0.100 |
| Hadleigh Marsh deionised water 0 hours to 24 hours | Blanks | 0.333 | 0.333 | 0.333 | 0.333 |
| | Samples | 0.400 | 0.100 | 0.100 | 0.100 |
| Hadleigh Marsh artificial seawater 0 hours to 24 hours | Blanks | 0.333 | 0.333 | 0.333 | 0.333 |
| | Samples | 0.400 | 0.100 | 0.100 | 0.100 |

## 4.15 Dissolved Organic Carbon in the leachates

Table 4.7 shows the results of the Dissolved Organ Carbon (DOC) analysis. The DOC concentrations show little variability with 88% of the samples being within 2 mg $L^{-1}$ of the median values. Comparing the DOC concentrations for the two landfill sites shows that, for both leachant types, the Hadleigh Marsh leachates had significantly higher DOC concentrations than the Leigh Marshes leachates (Mann-Whitney U tests, both leachants $p<0.001$). Comparing the DOC concentrations for the two leachant types shows that, for both landfill sites, the artificial seawater leachates contain significantly more DOC than the deionised water leachates (Mann-Whitney U tests, both sites $p<0.001$).

**Table 4.7: Dissolved Organic Carbon concentrations in the leachates (n=12)**

| | **Deionised water leaches (mg $L^{-1}$)** | | | | **Artificial seawater leaches (mg $L^{-1}$)** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Min.** | **Max.** | **Range** | **Median** | **Min.** | **Max.** | **Range** | **Median** |
| Leigh Marshes | 5.5 | 16 | 11 | 15 | 17 | 36 | 19 | 19 |
| Hadleigh Marshes | 25 | 42 | 17 | 38 | 39 | 42 | 3 | 41 |

## 4.16 Metal concentrations in the leachates

The concentrations of metals in the leachates at the end of the leaching experiments are shown in Figure 4.3. Metal concentrations in the leachates were highly variable with differences of up to three orders of magnitude between the minimum and maximum concentrations of some metals, e.g. Cd in the Hadleigh Marshes artificial seawater leachates, and between median concentrations of different metals, e.g. Co compared to Zn in the Hadleigh Marshes artificial seawater leachates.

Comparing the results of the deionised water leaching experiments (Mann-Whitney U tests) shows the concentrations of Al, Cr, Cu and Fe were significantly higher in the Hadleigh Marsh leachates compared to the Leigh Marshes leachates, and Co, Ni, Pb and Zn were significantly higher in the Leigh Marshes leachates compared to the Hadleigh Marsh leachates (see Table 4.8 for statistical data). The median concentrations of the metals were the same order of magnitude in the leachates for both landfill sites. However, the maximum concentration of Pb in the Leigh Marshes deionised water leachates was an order of magnitude higher than the maximum concentration of Pb in the Hadleigh Marsh deionised water leachates.

Comparing the results of the artificial seawater leaching experiments (Mann-Whitney U tests) shows the concentrations of Cd, Cr and Fe were significantly higher in the Hadleigh Marsh leachates compared to the Leigh Marshes leachates, and Co, Mn, Ni and Pb were higher in the Leigh Marshes leachates compared to the Hadleigh Marsh leachates (see Table 4.9 for statistical data). The median and maximum concentrations of Pb in the Leigh Marshes artificial seawater leachates were an order of magnitude higher than the median and maximum concentration of Pb in the Hadleigh Marsh artificial seawater leachates.

Comparing the results of the deionised water leaching experiments (Mann-Whitney U tests) to the results of the artificial seawater leaching experiments for each site shows the metal concentrations were significantly higher in the artificial seawater leachates than the deionised water leachates, with the exception of Al for the Hadleigh Marsh samples where there is no significant difference (see Table 4.10 and Table 4.11 for statistical data).

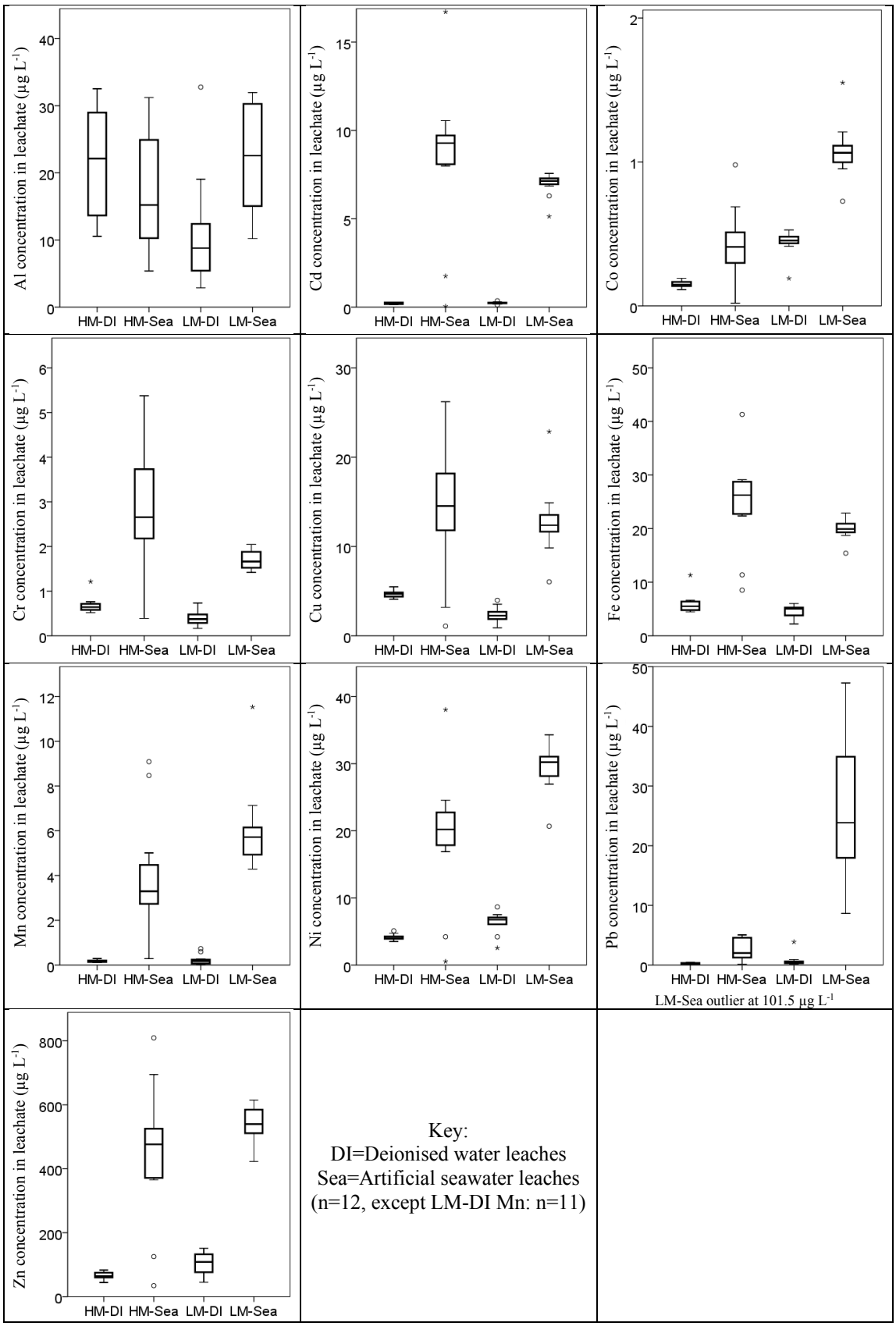



**Figure 4.3: Concentrations of metals in the leachates at the end of the 24 hour leaching experiments**

**Table 4.8: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the concentrations of metals in leachates when leaching samples from Hadleigh Marsh and Leigh Marshes landfills in deionised water (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (µg $L^{-1}$) (n=12, except Mn: n=11) | Hadleigh Marsh median (µg $L^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.004 | LM<HM | 8.78 | 22 |
| Cd | 0.386 | No | 0.24 | 0.22 |
| Co | <0.001 | LM>HM | 0.45 | 0.15 |
| Cr | 0.001 | LM<HM | 0.37 | 0.64 |
| Cu | <0.001 | LM<HM | 2.25 | 4.68 |
| Fe | 0.043 | LM<HM | 5.00 | 5.51 |
| Mn | 0.758 | No | 0.17 | 0.15 |
| Ni | 0.001 | LM>HM | 6.74 | 4.02 |
| Pb | 0.023 | LM>HM | 0.40 | 0.23 |
| Zn | 0.015 | LM>HM | 109 | 65 |

**Table 4.9: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the concentrations of metals in leachates when leaching samples from Hadleigh Marsh and Leigh Marshes landfills in seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (µg $L^{-1}$) (n=12) | Hadleigh Marsh median (µg $L^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.204 | No | 23 | 15 |
| Cd | 0.006 | LM<HM | 7.14 | 9.28 |
| Co | <0.001 | LM>HM | 1.06 | 0.41 |
| Cr | 0.006 | LM<HM | 1.66 | 2.66 |
| Cu | 0.273 | No | 13 | 15 |
| Fe | 0.007 | LM<HM | 20 | 26 |
| Mn | 0.007 | LM>HM | 5.72 | 3.30 |
| Ni | 0.001 | LM>HM | 30 | 20 |
| Pb | <0.001 | LM>HM | 23.9 | 2.02 |
| Zn | 0.050 | No | 540 | 476 |

**Table 4.10: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the concentrations of metals in leachates when leaching samples from Leigh Marshes landfill in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised water median (µg $L^{-1}$) (n=12, except Mn: n=11) | Artificial seawater median (µg $L^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.003 | DI<Sea | 8.78 | 23 |
| Cd | <0.001 | DI<Sea | 0.24 | 7.14 |
| Co | <0.001 | DI<Sea | 0.45 | 1.06 |
| Cr | <0.001 | DI<Sea | 0.37 | 1.66 |
| Cu | <0.001 | DI<Sea | 2.25 | 13 |
| Fe | <0.001 | DI<Sea | 5.00 | 20 |
| Mn | <0.001 | DI<Sea | 0.17 | 5.72 |
| Ni | <0.001 | DI<Sea | 6.74 | 30 |
| Pb | <0.001 | DI<Sea | 0.40 | 23.9 |
| Zn | <0.001 | DI<Sea | 109 | 540 |

**Table 4.11: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the concentrations of metals in leachates when leaching samples from Hadleigh Marsh landfill in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised water median (µg $L^{-1}$) (n=12) | Artificial seawater median (µg $L^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.149 | No | 22 | 15 |
| Cd | 0.001 | DI<Sea | 0.22 | 9.28 |
| Co | 0.005 | DI<Sea | 0.15 | 0.41 |
| Cr | 0.001 | DI<Sea | 0.64 | 2.66 |
| Cu | 0.006 | DI<Sea | 4.68 | 15 |
| Fe | <0.001 | DI<Sea | 5.51 | 26 |
| Mn | <0.001 | DI<Sea | 0.15 | 3.30 |
| Ni | 0.001 | DI<Sea | 4.02 | 20 |
| Pb | <0.001 | DI<Sea | 0.23 | 2.02 |
| Zn | 0.001 | DI<Sea | 65 | 476 |

## 4.17 Initial concentrations of metals in the matrix material

The initial (pre-leaching) concentrations of metals within the matrix material, shown in Table 4.12 and Table 4.13, were determined indirectly by mass-balance calculations using leachate concentration data (Section 4.16), post-leaching moisture content (Appendix B), and metal concentrations in the matrix material post-leaching (Appendix B). Initial concentrations of each of the metals in the matrix material vary by up to an order of magnitude between samples. Al and Fe were the most abundant metals present in both Hadleigh Marsh and Leigh Marshes matrix materials, and were present in concentrations an order of magnitude higher than other metals.

Comparing the calculated initial concentrations of metals in the matrix material used for the deionised water leaches (Mann-Whitney U tests), the Leigh Marshes samples had significantly higher concentrations of Cd, Co, Cr, Cu, Fe, Mn, Ni and Pb compared to the Hadleigh Marsh samples (see Table 4.14 for statistical data). Comparing the initial concentrations of metals in the matrix material used for the artificial seawater leaches, the Leigh Marshes samples had significantly higher concentrations of Cd, Co, Cu, Fe, Mn, Ni and Pb compared to the Hadleigh Marsh samples (see Table 4.15 for statistical data).

There were no significant differences in initial metal concentrations between the matrix material used for the Leigh Marshes deionised water and artificial seawater leaches. There were significantly higher Cr, Fe and Ni concentrations in the matrix material used for the Hadleigh Marsh deionised water leachates compared to the artificial seawater leaches ($p=0.002$, 0.002 and 0.007 respectively, see Table 4.13 for other statistical data).

**Table 4.12: Calculated initial (pre-leaching) concentrations of metals on matrix material from Leigh Marshes landfill**

| Element | Initial metal (mg) per kg of dried matrix material | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Samples for deionised water (n=12) | | | | Samples for artificial seawater (n=12) | | | |
| | Min. | Max. | Range | Median | Min. | Max. | Range | Median |
| Al | 10686 | 14537 | 3851 | 12123 | 11210 | 14485 | 3275 | 12161 |
| Cd | 2.09 | 3.01 | 0.92 | 2.65 | 2.28 | 26.76 | 24 | 2.85 |
| Co | 29 | 75 | 47 | 46 | 34 | 66 | 32 | 41 |
| Cr | 22 | 107 | 85 | 48 | 23 | 184 | 161 | 46 |
| Cu | 439 | 4327 | 3889 | 652 | 415 | 841 | 426 | 524 |
| Fe | 68474 | 89078 | 20604 | 79405 | 57837 | 78122 | 20285 | 72916 |
| Mn | 684 | 3884 | 3200 | 959 | 722 | 1201 | 478 | 830 |
| Ni | 110 | 234 | 124 | 154 | 112 | 206 | 94 | 141 |
| Pb | 981 | 4128 | 3147 | 1383 | 901 | 2182 | 1281 | 1318 |
| Zn | 1088 | 2367 | 1279 | 1334 | 974 | 1567 | 593 | 1299 |

**Table 4.13: Calculated initial (pre-leaching) concentrations of metals on matrix material from Hadleigh Marsh landfill**

| Element | Initial metal (mg) per kg of dried matrix material | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Samples for deionised water (n=12) | | | | Samples for artificial seawater (n=12) | | | |
| | Min. | Max. | Range | Median | Min. | Max. | Range | Median |
| Al | 11094 | 12484 | 1390 | 11589 | 11081 | 12745 | 1664 | 11620 |
| Cd | 1.42 | 2.13 | 0.70 | 1.74 | 1.53 | 1.97 | 0.44 | 1.83 |
| Co | 16 | 21 | 4.8 | 18 | 15 | 19 | 3.8 | 17 |
| Cr | 56 | 78 | 22 | 66 | 52 | 68 | 16 | 60 |
| Cu | 92 | 205 | 112 | 121 | 91 | 188 | 97 | 122 |
| Fe | 35601 | 41816 | 6216 | 39477 | 34365 | 38063 | 3698 | 36484 |
| Mn | 427 | 544 | 117 | 471 | 418 | 518 | 100 | 470 |
| Ni | 48 | 62 | 14 | 57 | 46 | 59 | 12 | 51 |
| Pb | 397 | 810 | 413 | 585 | 417 | 744 | 327 | 562 |
| Zn | 1107 | 1872 | 765 | 1533 | 1107 | 1720 | 613 | 1471 |

**Table 4.14: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the calculated initial metal concentrations in samples from Hadleigh Marsh and Leigh Marshes landfills used for leaching in deionised water (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (mg $kg^{-1}$) (n=12) | Hadleigh Marsh median (mg $kg^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.248 | No | 12123 | 11589 |
| Cd | <0.001 | LM>HM | 2.65 | 1.74 |
| Co | <0.001 | LM>HM | 46 | 18 |
| Cr | 0.028 | LM<HM | 48 | 66 |
| Cu | <0.001 | LM>HM | 652 | 121 |
| Fe | <0.001 | LM>HM | 79405 | 39477 |
| Mn | <0.001 | LM>HM | 959 | 471 |
| Ni | <0.001 | LM>HM | 154 | 57 |
| Pb | <0.001 | LM>HM | 1383 | 585 |
| Zn | 0.204 | No | 1334 | 1533 |

**Table 4.15: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the calculated initial metal concentrations in samples from Hadleigh Marsh and Leigh Marshes landfills used for leaching in artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (mg $kg^{-1}$) (n=12) | Hadleigh Marsh median (mg $kg^{-1}$) (n=12) |
|---|---|---|---|---|
| Al | 0.057 | No | 12161 | 11620 |
| Cd | <0.001 | LM>HM | 2.85 | 1.83 |
| Co | <0.001 | LM>HM | 41 | 17 |
| Cr | 0.184 | No | 46 | 60 |
| Cu | <0.001 | LM>HM | 524 | 122 |
| Fe | <0.001 | LM>HM | 72916 | 36484 |
| Mn | <0.001 | LM>HM | 830 | 470 |
| Ni | <0.001 | LM>HM | 141 | 51 |
| Pb | <0.001 | LM>HM | 1318 | 562 |
| Zn | 0.094 | No | 1299 | 1471 |

## 4.18 Proportion of metals released from the matrix material

The proportion of each metal released during the leaching experiment as a percentage of the initial concentration of that metal in the sample prior to leaching is shown in Figure 4.4 for each leachant type by landfill site. The proportions of metals released were highly variable, but with the exception of Cd in artificial seawater, maximum proportions of less than 1% of the metals were released. In the artificial seawater leaches, Cd had a maximum released proportion of 3.2% for Leigh Marshes and 8.8% for Hadleigh Marsh.

Comparing the proportion of metals released from the matrix material in the deionised water leaching experiments (Mann-Whitney U tests) shows there were significant differences between sites. Greater proportions of Al, Cd, Cu, Fe, Mn and Ni were released from the Hadleigh Marsh matrix material compared to the Leigh Marshes matrix material, and a greater proportion of Zn was released from the Leigh Marshes matrix material compared to the Hadleigh Marsh matrix material (see Table 4.16 for statistical data). The median proportion of Cu released from the Hadleigh Marsh matrix material in the deionised water leaching experiments was an order of magnitude higher than the median proportion of Cu released from the Leigh Marshes matrix material.

Comparing the proportions of metals released from the matrix material in the artificial seawater leaching experiments (Mann-Whitney U tests) also shows there were significant differences between sites. Greater proportions of Cd, Cu, Fe and Ni were released from the Hadleigh Marsh matrix material compared to the Leigh Marshes matrix material, and greater proportions of Pb and Zn were released from the Leigh Marshes matrix material compared to the Hadleigh Marsh matrix material (see Table 4.17 for statistical data). The median proportion of Cu released from the Hadleigh Marsh matrix material in the artificial seawater leaching experiments was an order of magnitude higher than the median proportion of Cu released from the Leigh Marshes matrix material. The median proportion of Pb released from the Leigh Marshes matrix material in the artificial seawater leaching experiments was an order of magnitude higher than the median proportion of Pb released from the Hadleigh Marsh matrix material.

Comparing the proportions of metals released into the two leachant types shows there were significant differences between the proportions of metals released into deionised water and artificial seawater. For both sites significantly higher proportions of the metals were released in artificial seawater compared to deionised water, with the exception of Al in the Hadleigh Marsh samples for which there is no significant difference between the proportions released in deionised water and artificial seawater (Table 4.18 and Table 4.19). The differences in the median proportions of metals released were up to two orders of magnitude between the leachant types.

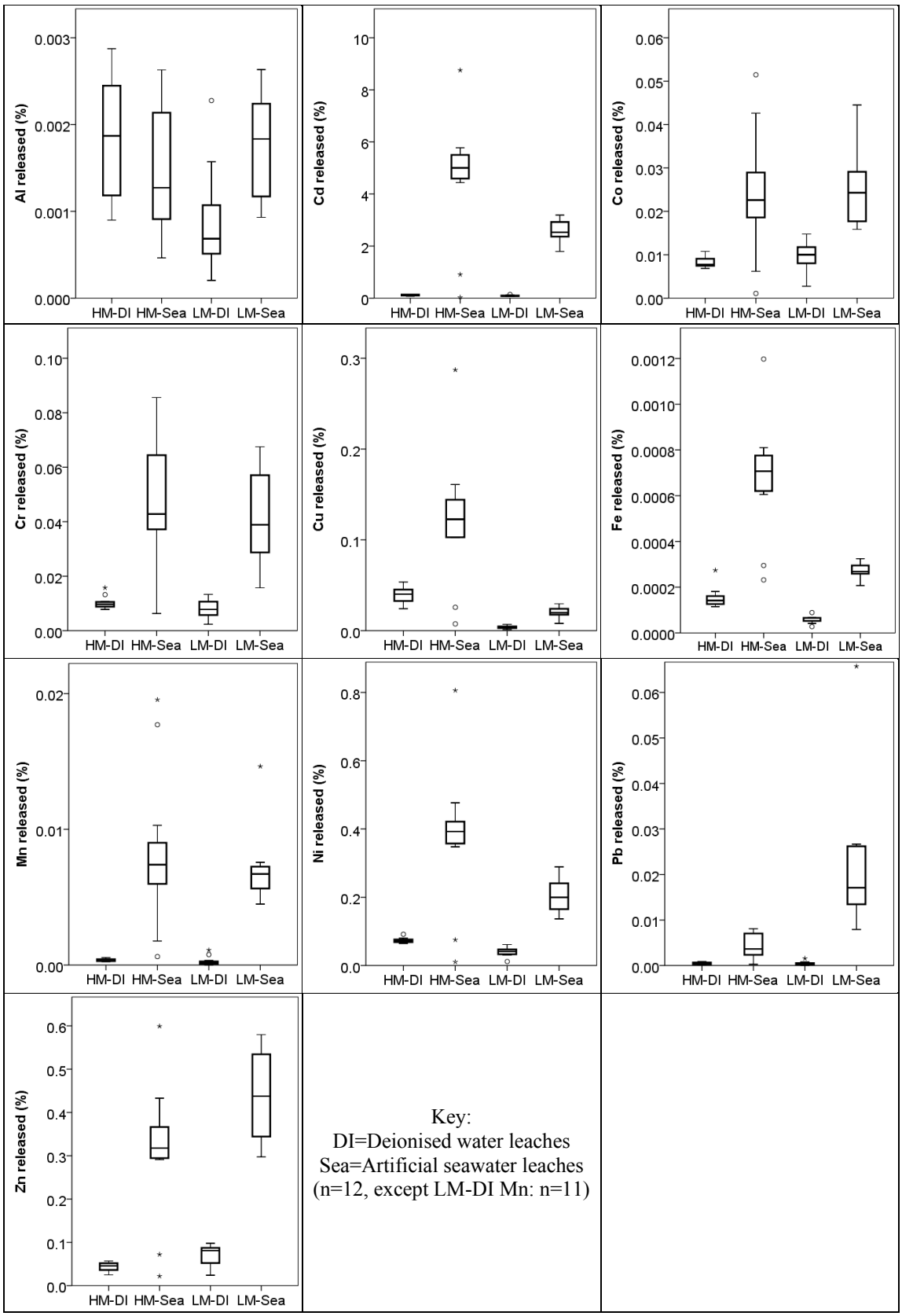



**Figure 4.4: Proportion of metals released from the Hadleigh Marsh (HM) and Leigh Marshes (LM) matrix material during the leaching experiments as a percentage of the initial concentration of each metal in the sample**

**Table 4.16: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the proportions of metals released between samples from Hadleigh Marsh and Leigh Marshes landfills when leached in deionised water (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (%) (n=12, except Mn: n=11) | Hadleigh Marsh median (%) (n=12) |
|---|---|---|---|---|
| Al | 0.001 | LM<HM | 0.00069 | 0.00187 |
| Cd | 0.013 | LM<HM | 0.08951 | 0.12726 |
| Co | 0.106 | No | 0.01001 | 0.00779 |
| Cr | 0.073 | No | 0.00781 | 0.00970 |
| Cu | <0.001 | LM<HM | 0.00350 | 0.04025 |
| Fe | <0.001 | LM<HM | 0.00006 | 0.00014 |
| Mn | 0.031 | LM<HM | 0.00015 | 0.00031 |
| Ni | <0.001 | LM<HM | 0.04116 | 0.07162 |
| Pb | 0.525 | No | 0.00031 | 0.00034 |
| Zn | 0.006 | LM>HM | 0.08081 | 0.04553 |

**Table 4.17: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the proportions of metals released between samples from Hadleigh Marsh and Leigh Marshes landfills when leached in seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (%) (n=12) | Hadleigh Marsh median (%) (n=12) |
|---|---|---|---|---|
| Al | 0.248 | No | 0.00183 | 0.00127 |
| Cd | 0.006 | LM<HM | 2.52865 | 5.00197 |
| Co | 0.908 | No | 0.02430 | 0.02260 |
| Cr | 0.488 | No | 0.03887 | 0.04283 |
| Cu | 0.001 | LM<HM | 0.01984 | 0.12271 |
| Fe | 0.001 | LM<HM | 0.00027 | 0.00071 |
| Mn | 0.326 | No | 0.00671 | 0.00739 |
| Ni | 0.006 | LM<HM | 0.19961 | 0.39256 |
| Pb | <0.001 | LM>HM | 0.01710 | 0.00363 |
| Zn | 0.028 | LM>HM | 0.43749 | 0.31765 |

**Table 4.18: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the proportions of metals released between Leigh Marshes landfill samples leached in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised water median (%) (n=12, except Mn: n=11) | Artificial seawater median (%) (n=12) |
|---|---|---|---|---|
| Al | 0.002 | DI<Seawater | 0.00069 | 0.00183 |
| Cd | <0.001 | DI<Seawater | 0.08951 | 2.52865 |
| Co | <0.001 | DI<Seawater | 0.01001 | 0.02430 |
| Cr | <0.001 | DI<Seawater | 0.00781 | 0.03887 |
| Cu | <0.001 | DI<Seawater | 0.00350 | 0.01984 |
| Fe | <0.001 | DI<Seawater | 0.00006 | 0.00027 |
| Mn | <0.001 | DI<Seawater | 0.00015 | 0.00671 |
| Ni | <0.001 | DI<Seawater | 0.04116 | 0.19961 |
| Pb | <0.001 | DI<Seawater | 0.00031 | 0.01710 |
| Zn | <0.001 | DI<Seawater | 0.08081 | 0.43749 |

**Table 4.19: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in the proportions of metals released between Hadleigh Marsh landfill samples leached in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised water median (%) (n=12) | Artificial seawater median (%) (n=12) |
|---|---|---|---|---|
| Al | 0.166 | No | 0.00187 | 0.00127 |
| Cd | 0.001 | DI<Seawater | 0.12726 | 5.00197 |
| Co | 0.006 | DI<Seawater | 0.00779 | 0.02260 |
| Cr | 0.001 | DI<Seawater | 0.00970 | 0.04283 |
| Cu | 0.005 | DI<Seawater | 0.04025 | 0.12271 |
| Fe | <0.001 | DI<Seawater | 0.00014 | 0.00071 |
| Mn | <0.001 | DI<Seawater | 0.00031 | 0.00739 |
| Ni | 0.001 | DI<Seawater | 0.07162 | 0.39256 |
| Pb | <0.001 | DI<Seawater | 0.00034 | 0.00363 |
| Zn | 0.001 | DI<Seawater | 0.04553 | 0.31765 |

## Correlation analyses

To explore the relationship between the proportions of metals released and DOC Spearman's rank correlation coefficients were calculated (Table 4.20, Table 4.21, Table 4.22 and Table 4.23). Few correlations were found between the proportions of metals released in the deionised water leachates, but numerous correlations were found between the proportions of metals released in the artificial seawater leachates. The correlations between the proportions of metals released were all positive.

Only the proportion of Mn released correlated with DOC in the Leigh Marshes deionised water leachates. The proportions released of several metals correlated with DOC in the Leigh Marshes artificial seawater leachates. The correlations between the proportions of released metals and DOC were all negative. There were no correlations between proportions of metals released and DOC in the Hadleigh Marsh leachates.

**Table 4.20: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the deionised water leachants and DOC for the Leigh Marshes matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | | proportion of metal released | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | DOC | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
| | DOC | | | | | | | | -0.780 | | | |
| proportion of metal released | Al | | | | | | | | | | 0.615 | **0.888** |
| | Cd | | | | | | | 0.601 | | | | |
| | Co | | | | | | | | | **0.839** | | |
| | Cr | | | | | | | | | | | |
| | Cu | | | | | | | | | | | |
| | Fe | | | | | | | | | | | |
| | Mn | | | | | | | | | | | |
| | Ni | | | | | | | | | | | |
| | Pb | | | | | | | | | | | |
| | Zn | | | | | | | | | | | |

**Table 4.21: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the artificial seawater leachants and DOC for the Leigh Marshes matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | | proportion of metal released | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | DOC | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
| | DOC | | | | -0.651 | | | | -0.690 | -0.581 | -0.725 | |
| proportion of metal released | Al | | | | | | | | | | | |
| | Cd | | | | 0.636 | | | **0.881** | 0.629 | 0.678 | | |
| | Co | | | | | 0.706 | | 0.643 | **0.846** | **0.972** | 0.678 | 0.769 |
| | Cr | | | | | | | | 0.685 | 0.713 | | 0.776 |
| | Cu | | | | | | | | | | | |
| | Fe | | | | | | | | 0.580 | 0.706 | | |
| | Mn | | | | | | | | | **0.804** | | 0.664 |
| | Ni | | | | | | | | | | 0.692 | 0.783 |
| | Pb | | | | | | | | | | | 0.727 |
| | Zn | | | | | | | | | | | |

**Table 4.22: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the deionised water leachants and DOC for the Hadleigh Marsh matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | | proportion of metal released | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | DOC | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
| | DOC | | | | | | | | | | | |
| proportion of metal released | Al | | | | | | | | | 0.755 | 0.657 | 0.741 |
| | Cd | | | | | | | | | | | |
| | Co | | | | | | | | | | | |
| | Cr | | | | | | | | | | | |
| | Cu | | | | | | | | | **0.825** | | 0.636 |
| | Fe | | | | | | | | | | | |
| | Mn | | | | | | | | | | | |
| | Ni | | | | | | | | | | 0.755 | 0.783 |
| | Pb | | | | | | | | | | | **0.839** |
| | Zn | | | | | | | | | | | |

**Table 4.23: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the artificial seawater leachants and DOC for the Hadleigh Marsh matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | | proportion of metal released | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | DOC | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
| | DOC | | | | | | | | | | | |
| proportion of metal released | Al | | | 0.650 | | | 0.783 | | 0.790 | | **0.860** | |
| | Cd | | | | | | 0.706 | | | 0.594 | 0.699 | 0.797 |
| | Co | | | | | **0.874** | | **0.832** | 0.713 | 0.594 | | |
| | Cr | | | | | | 0.650 | **0.846** | | | | |
| | Cu | | | | | | | | 0.671 | 0.580 | 0.748 | 0.643 |
| | Fe | | | | | | | | 0.762 | 0.797 | 0.678 | 0.797 |
| | Mn | | | | | | | | | 0.713 | **0.860** | 0.783 |
| | Ni | | | | | | | | | | 0.664 | **0.881** |
| | Pb | | | | | | | | | | | **0.825** |
| | Zn | | | | | | | | | | | |

Correlation analyses (Spearman's rank) were also carried out to explore the association between the proportions of metals released and the initial metal concentrations in the matrix material (Table 4.24, Table 3.17, Table 4.26 and Table 4.27). For Leigh Marshes, correlations between the proportions of metals released and the initial metal concentrations were more numerous in the artificial seawater leachates than the deionised water leachates. For Hadleigh Marsh, correlations between the proportions of metals released and the initial metal concentrations were more numerous in the deionised water leachates than the artificial seawater leachates. All of the correlations between the proportions of metals released and the initial concentrations of metals were negative.

**Table 4.24: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the deionised water leachants and the initial concentrations in the Leigh Marshes matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | initial concentration of metal in the matrix material | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Al** | **Cd** | **Co** | **Cr** | **Cu** | **Fe** | **Mn** | **Ni** | **Pb** | **Zn** |
| **proportion of metal released** | **Al** | | | | | | | | | | |
| | **Cd** | | | | | | | | | | |
| | **Co** | -0.727 | **-0.811** | **-0.909** | **-0.902** | | | **-0.818** | **-0.895** | | -0.587 |
| | **Cr** | | | | | -0.650 | | | | | |
| | **Cu** | | | | | | | | | | |
| | **Fe** | | | | | | | | | | |
| | **Mn** | | | | | | | | | | |
| | **Ni** | | -0.643 | -0.713 | -0.755 | | | -0.594 | -0.685 | | |
| | **Pb** | | | | | | | | | | |
| | **Zn** | | | | | | | | | | |

**Table 4.25: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the artificial seawater leachants and the initial concentrations in the Leigh Marshes matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | initial concentration of metal in the matrix material | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Al** | **Cd** | **Co** | **Cr** | **Cu** | **Fe** | **Mn** | **Ni** | **Pb** | **Zn** |
| **proportion of metal released** | **Al** | | | | | | | | | | |
| | **Cd** | | **-0.916** | | | | | | | | |
| | **Co** | -0.734 | -0.650 | **-0.867** | -0.727 | | | -0.727 | **-0.853** | | |
| | **Cr** | **-0.909** | | **-0.853** | **-0.888** | | | -0.671 | -0.797 | | -0.755 |
| | **Cu** | | | | | | | | | | |
| | **Fe** | | **-0.888** | -0.615 | | | | | -0.629 | | |
| | **Mn** | -0.643 | -0.657 | -0.664 | | | | | -0.643 | | |
| | **Ni** | -0.734 | -0.657 | **-0.923** | -0.748 | | | -0.678 | **-0.867** | | -0.678 |
| | **Pb** | | | | | | | | -0.629 | | |
| | **Zn** | -0.657 | -0.580 | -0.741 | -0.769 | | | | -0.734 | | **-0.846** |

**Table 4.26: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the deionised water leachants and the initial concentrations in the Hadleigh Marsh matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | initial concentration of metal in the matrix material | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Al** | **Cd** | **Co** | **Cr** | **Cu** | **Fe** | **Mn** | **Ni** | **Pb** | **Zn** |
| **proportion of metal released** | **Al** | | | -0.657 | | | -0.587 | -0762 | | | |
| | **Cd** | | | | | | | | | | |
| | **Co** | | | | | | | | | | |
| | **Cr** | | | | | | | | | | |
| | **Cu** | | -0.650 | -0.587 | -0.608 | **-0.874** | | -0.776 | -0.601 | | -0.601 |
| | **Fe** | | | | | | | | | | |
| | **Mn** | | | | | | | | | | |
| | **Ni** | | | -0.580 | | -0.713 | | **-0.811** | | | -0.657 |
| | **Pb** | | | | | | | -0.699 | | | -0.615 |
| | **Zn** | | | | | | | | | | -0.671 |

**Table 4.27: Spearman's rank coefficients for analysis of correlations between the proportions of metals released into the artificial seawater leachants and the initial concentrations in the Hadleigh Marsh matrix material. Only values with a significance level alpha=0.05 are shown. Underlined values show $r_s$>0.8**

| | | initial concentration of metal in the matrix material | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Al** | **Cd** | **Co** | **Cr** | **Cu** | **Fe** | **Mn** | **Ni** | **Pb** | **Zn** |
| **proportion of metal released** | **Al** | | | | | | | | | | |
| | **Cd** | | | -0.664 | -0.636 | -0.734 | -0.713 | -0.608 | -0.657 | | |
| | **Co** | -0.657 | | | | | | | | | |
| | **Cr** | -0.734 | | | | | | | | | |
| | **Cu** | | | | | -0.580 | | | | | |
| | **Fe** | **-0.881** | | | | | | | | | |
| | **Mn** | | | | | | | | | | |
| | **Ni** | -0.692 | | | | | | | | | |
| | **Pb** | | | | | | | | | | |
| | **Zn** | -0.692 | | | | | -0.713 | | | | |

Redox potential, pH, temperature and DO sample sizes were too small (n=3 for each of the landfill site/leachant combinations) for them to be included in the correlation analyses.

## 4.19 Mass of metal released per kilogram of dry waste

The Landfill Directive and BS EN 12457-2 require that the leaching results are reported as the mass of metals released per kilogram of dry waste to enable comparison to Waste Acceptance Criteria limit values. The data are presented in the requisite format for Leigh Marshes in Table 4.28 and for Hadleigh Marsh in Table 4.29.

Comparing the mass of metals released per kilogram of dry matrix material for the two landfill sites (Mann-Whitney U tests) in the same leachant type shows there were significant differences between the two sites. In the deionised water leaches Leigh Marshes matrix material released more Co, Ni, Pb and Zn than the Hadleigh Marshes matrix material, but the Hadleigh Marsh matrix material released more Al, Cr, Cu and Fe than the Leigh Marshes matrix material (see Table 4.30 for statistical data). In the artificial seawater leaches Leigh Marshes matrix material released more Co, Mn, Ni, Pb and Zn than the Hadleigh Marshes matrix material, but the Hadleigh Marsh matrix material released more Cd, Cr and Fe than the Leigh Marshes matrix material (see Table 4.31 for statistical data).

Comparing the mass of metals released per kilogram of dry matrix material into the two leachant types (Mann-Whitney U tests) shows significantly higher masses of each metal were released in artificial seawater compared to deionised water per kilogram of matrix material, with the exception of Al in the Hadleigh Marsh leaches where there is no significant difference between the deionised water and artificial seawater leaches (see Table 4.32 and Table 4.33 for statistical data).

**Table 4.28: Mass of metals released during leaching (µg) per kilogram of dried matrix material from Leigh Marshes landfill**

| | Mass of metal released (µg) per kg of dried matrix material | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Leached in deionised water (n=12; except Mn: n=11) | | | | Leached in artificial seawater (n=12) | | | |
| **Element** | **Min.** | **Max.** | **Range** | **Median** | **Min.** | **Max.** | **Range** | **Median** |
| Al | 28 | 331 | 303 | 89 | 102 | 320 | 218 | 228 |
| Cd | 1.3 | 3.5 | 2.2 | 2.5 | 51 | 77 | 26 | 72 |
| Co | 1.9 | 5.3 | 3.4 | 4.5 | 7.3 | 16 | 8.5 | 11 |
| Cr | 1.7 | 7.3 | 5.6 | 3.8 | 14 | 21 | 6.4 | 17 |
| Cu | 8.9 | 40 | 31 | 22 | 60 | 233 | 173 | 124 |
| Fe | 22 | 61 | 39 | 50 | 154 | 230 | 76 | 201 |
| Mn | 0.2 | 7.5 | 7.3 | 1.7 | 43 | 118 | 75 | 58 |
| Ni | 25 | 88 | 62 | 67 | 207 | 350 | 143 | 301 |
| Pb | 1.4 | 39 | 38 | 4.1 | 87 | 1036 | 949 | 239 |
| Zn | 447 | 1529 | 1083 | 1088 | 4180 | 6270 | 2090 | 5395 |

**Table 4.29: Mass of metals released during leaching (µg) per kilogram of dried matrix material from Hadleigh Marsh landfill**

| | Metal released (µg) per kg of dried matrix material | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Leached in deionised water (n=12) | | | | Leached in artificial seawater (n=12) | | | |
| **Element** | **Min.** | **Max.** | **Range** | **Median** | **Min.** | **Max.** | **Range** | **Median** |
| Al | 105 | 323 | 219 | 222 | 54 | 304 | 250 | 150 |
| Cd | 1.4 | 2.7 | 1.2 | 2.2 | 0.5 | 166 | 166 | 92 |
| Co | 1.2 | 1.9 | 0.7 | 1.5 | 0.2 | 9.9 | 9.7 | 4.0 |
| Cr | 5.3 | 12 | 7.0 | 6.5 | 3.9 | 54 | 50 | 26 |
| Cu | 41 | 55 | 14 | 46 | 11 | 261 | 250 | 143 |
| Fe | 44 | 115 | 70 | 56 | 86 | 412 | 325 | 262 |
| Mn | 1.1 | 3.0 | 1.9 | 1.6 | 2.9 | 92 | 89 | 32 |
| Ni | 35 | 51 | 17 | 40 | 5.4 | 379 | 374 | 196 |
| Pb | 1.3 | 4.4 | 3.0 | 2.3 | 1.6 | 50 | 49 | 20 |
| Zn | 433 | 827 | 394 | 651 | 349 | 8061 | 7712 | 4682 |

**Table 4.30: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in metals released per kg of matrix material for Leigh Marshes and Hadleigh Marsh when leached in deionised water (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (n=12, except Mn: n=11) | Hadleigh Marsh median (n=12) |
|---|---|---|---|---|
| | | | (µg released per kg of waste) | |
| Al | 0.003 | LM<HM | 89.5 | 221.9 |
| Cd | 0.386 | No | 2.48 | 2.24 |
| Co | <0.001 | LM>HM | 4.54 | 1.47 |
| Cr | 0.001 | LM<HM | 3.75 | 6.46 |
| Cu | <0.001 | LM<HM | 22.5 | 46.2 |
| Fe | 0.038 | LM<HM | 50.3 | 55.6 |
| Mn | 0.712 | No | 1.35 | 1.58 |
| Ni | 0.001 | LM>HM | 67.4 | 39.9 |
| Pb | 0.015 | LM>HM | 4.08 | 2.30 |
| Zn | 0.018 | LM>HM | 1088 | 651.3 |

**Table 4.31: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in metals released per kg of matrix material for Leigh Marshes and Hadleigh Marsh when leached in artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Leigh Marshes median (n=12) | Hadleigh Marsh median (n=12) |
|---|---|---|---|---|
| | | | (µg released per kg of waste) | |
| Al | 0.119 | No | 227.8 | 149.6 |
| Cd | 0.006 | LM<HM | 71.6 | 92.5 |
| Co | <0.001 | LM>HM | 10.70 | 4.03 |
| Cr | 0.006 | LM<HM | 16.59 | 25.8 |
| Cu | 0.299 | No | 124.46 | 142.7 |
| Fe | 0.011 | LM<HM | 200.84 | 261.9 |
| Mn | 0.007 | LM>HM | 57.56 | 32.2 |
| Ni | 0.001 | LM>HM | 301.0 | 196.1 |
| Pb | <0.001 | LM>HM | 238.83 | 20.1 |
| Zn | 0.043 | LM>HM | 5395.5 | 4681.8 |

**Table 4.32: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in metals released per kg of Leigh Marshes matrix material when leached in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised median (n=12, except Mn: n=11) | Sea median (n=12) |
|---|---|---|---|---|
| | | | (µg released per kg of waste) | |
| Al | 0.003 | Deionised<Sea | 89.5 | 227.8 |
| Cd | <0.001 | Deionised<Sea | 2.48 | 71.6 |
| Co | <0.001 | Deionised<Sea | 4.54 | 10.70 |
| Cr | <0.001 | Deionised<Sea | 3.75 | 16.59 |
| Cu | <0.001 | Deionised<Sea | 22.5 | 124.46 |
| Fe | <0.001 | Deionised<Sea | 50.3 | 200.84 |
| Mn | <0.001 | Deionised<Sea | 1.35 | 57.56 |
| Ni | <0.001 | Deionised<Sea | 67.4 | 301.0 |
| Pb | <0.001 | Deionised<Sea | 4.08 | 238.83 |
| Zn | <0.001 | Deionised<Sea | 1088 | 5395.5 |

**Table 4.33: Results of the Mann-Whitney U tests applied to determine whether there were significant differences in metals released per kg of Hadleigh Marsh matrix material when leached in deionised water and artificial seawater (critical p value = 0.050)**

| Element | Mann-Whitney U p value | Significant difference | Deionised median (n=12) | Sea median (n=12) |
|---|---|---|---|---|
| | | | (µg released per kg of waste) | |
| Al | 0.133 | No | 221.9 | 149.6 |
| Cd | 0.001 | Deionised<Sea | 2.24 | 92.5 |
| Co | 0.005 | Deionised<Sea | 1.47 | 4.03 |
| Cr | 0.001 | Deionised<Sea | 6.46 | 25.8 |
| Cu | 0.006 | Deionised<Sea | 46.2 | 142.7 |
| Fe | <0.001 | Deionised<Sea | 55.6 | 261.9 |
| Mn | <0.001 | Deionised<Sea | 1.58 | 32.2 |
| Ni | 0.001 | Deionised<Sea | 39.9 | 196.1 |
| Pb | <0.001 | Deionised<Sea | 2.30 | 20.1 |
| Zn | 0.001 | Deionised<Sea | 651.3 | 4681.8 |

## 4.20 Potential concentrations of metals in the receiving waters

To enable comparison to EQSs for the Protection of Surface Water Quality, concentrations of metals in the receiving waters as a result of metal release from eroded waste were calculated. The rate of erosion would influence the maximum metal concentrations in the receiving waters, but currently there are no suitable modelling tools for assessing landfill waste erosion rates due to the complex mechanical properties of waste being poorly understood (Dixon and Jones, 2005). Therefore, the worst case scenario of a landfill site failing and releasing all of its soluble metals in less than a day was assessed. In Chapter 3 it was estimated that 8,200-10,900 tonnes (dry mass) of the matrix material would be released if a waste cell from Hadleigh Marsh landfill were to erode, and approximately 204,500-272,500 tonnes (dry mass) of the matrix material would be released if Leigh Marshes landfill were to erode. These masses and the mass of metals released per kilogram of waste, determined from the leaching experiments, were substituted into Equation 4.10 to calculate the total mass of metals that could be released from each of the landfills if eroded into river water or seawater, based on the findings of the deionised water and artificial seawater leaching experiments respectively. The results of the calculations are shown in Table 4.34 for Leigh Marshes and in Table 4.35 for Hadleigh Marsh. The potential concentrations of the metals in the receiving waters were then calculated (as per Equation 4.9) using the Thames Estuary's tidal prism volume of 585,000,000 $m^3$ (Mikhailov and Mikhailova, 2012) as the volume available for dilution of the contaminants. The results of the calculations are shown in Table 4.36 for Leigh Marshes and in Table 4.37 for Hadleigh Marsh.

**Table 4.34: Total masses of metals that could be released from the matrix material into solution if Leigh Marshes landfill erodes based on median metal concentrations**

| Element | Total mass of metal released from matrix material if the site erodes (kg) | |
|---|---|---|
| | Eroded into river water (based on deionised water experiments) | Eroded into seawater (based on artificial seawater experiments) |
| Al | 18.3 - 24.4 | 46.6 - 62.1 |
| Cd | 0.51 - 0.68 | 14.6 - 19.5 |
| Co | 0.93 - 1.24 | 2.19 - 2.92 |
| Cr | 0.77 - 1.02 | 3.39 - 4.52 |
| Cu | 4.6 - 6.13 | 25.5 - 33.9 |
| Fe | 10.3 - 13.7 | 41.1 - 54.7 |
| Mn | 0.28 - 0.37 | 11.8 - 15.7 |
| Ni | 13.8 - 18.4 | 61.6 - 82.0 |
| Pb | 0.83 - 1.11 | 48.8 - 65.1 |
| Zn | 223 - 296 | 1,103 – 1,470 |

**Table 4.35: Total masses of metals that could be released from the matrix material into solution if a single waste cell from Hadleigh Marsh landfill erodes based on median metal concentrations**

| Element | Total mass of metal released from matrix material if a single waste cell erodes (kg) | |
|---|---|---|
| | Eroded into river water (based on deionised water experiments) | Eroded into seawater (based on artificial seawater experiments) |
| Al | 1.82 - 2.42 | 1.23 - 1.63 |
| Cd | 0.018 - 0.024 | 0.76 - 1.01 |
| Co | 0.01 - 0.02 | 0.03 - 0.04 |
| Cr | 0.05 - 0.07 | 0.21 - 0.28 |
| Cu | 0.38 - 0.50 | 1.17 - 1.56 |
| Fe | 0.46 - 0.61 | 2.15 - 2.85 |
| Mn | 0.01 - 0.02 | 0.26 - 0.35 |
| Ni | 0.33 - 0.43 | 1.61 - 2.14 |
| Pb | 0.02 - 0.03 | 0.16 - 0.22 |
| Zn | 5.34 - 7.10 | 38.4 - 51.04 |

**Table 4.36: Maximum dissolved metal concentrations in water resulting from the erosion of Leigh Marshes landfill in a single tidal cycle, based on a tidal prism volume of 585,000,000 $m^3$, compared to surface water quality EQSs**

| Element | Max. concentration when diluted by tidal prism volume (ng $L^{-1}$) | |
|---|---|---|
| | Eroded into river water | Eroded into seawater |
| Al | 41.7 | 106 |
| Cd | 1.16 | 33.4 |
| Co | 2.11 | 4.98 |
| Cr | 1.75 | 7.73 |
| Cu | 10.5 | 58.0 |
| Fe | 23.4 | 93.6 |
| Mn | 0.63 | 26.8 |
| Ni | 31.4 | 140 |
| Pb | 1.90 | 111 |
| Zn | 507 | 2,513 |

**Table 4.37: Maximum dissolved metal concentrations in water resulting from the erosion of a single Hadleigh Marsh waste cell in a single tidal cycle, based on a tidal prism volume of 585,000,000 $m^3$, compared to surface water quality EQSs**

| Element | Max. concentration when diluted by tidal prism volume (ng $L^{-1}$) | |
|---|---|---|
| | Eroded into river water | Eroded into seawater |
| Al | 4.13 | 2.79 |
| Cd | 0.04 | 1.72 |
| Co | 0.03 | 0.08 |
| Cr | 0.12 | 0.48 |
| Cu | 0.86 | 2.66 |
| Fe | 1.04 | 4.88 |
| Mn | 0.03 | 0.60 |
| Ni | 0.74 | 3.65 |
| Pb | 0.04 | 0.37 |
| Zn | 12.1 | 87.2 |

## 4.21 Proportion of metal released as a percentage of the initial concentration of the metal within the samples

Differences in the proportions of metals released from waste into solution in a leachant may be related to a number of factors relating to the physico-chemical properties of the waste and its metal content, e.g. initial metal load (Hatje et al., 2003), metal speciation and bonding (Bódog et al., 1996; Chaudhary and Banerjee, 2007; Flues et al., 2013), and the particle size, density, biodegradability and organic carbon content of the waste (Zhao et al., 2013; Mukherjee et al., 2014). The proportions of metals released from waste into a leachant may also be affected by the chemistry of the leachant, e.g. pH, redox potential (Baun and Christensen, 2004) and DOC concentrations (Kalbitz and Wennrich, 1998). In addition, chemical reactions between the waste and the leachant can affect the release of metals from the waste, e.g. formation of precipitates (Kjeldsen et al., 2002), formation of soluble complexes with anions in the leachant, and competition with cations in the leachant for sorption sites (Chapman and Wang, 2001; Acosta et al., 2011).

### 4.21.1 Comparison to proportions used in existing risk assessments

Although contaminant concentrations in leachates have been well studied, only Belevi and Baccini (1989) have previously investigated the proportion of metals released from inundated waste, and it is their results that are used to represent all landfills in existing landfill risk assessments (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013). The results of the deionised water leaching experiments using Leigh Marshes and Hadleigh Marsh matrix material are compared to data from Belevi and Baccini (1989) in Table 4.38 and Table 4.39 respectively. Belevi and Baccini (1989) only experimented using distilled water, they did not consider saline leachants. The proportion of Cd released in the Leigh Marshes and Hadleigh Marsh leaches is comparable to the proportion released in Belevi and Baccini's study, but the proportions of Cu, Fe, Pb and Zn released in their study were at least an order of magnitude higher. However, Belevi and Baccini dried and ground their samples prior to leaching, which can significantly increase metal mobility (Hu et al., 2008).

**Table 4.38: Proportion of metals released from Leigh Marshes landfill matrix material during the leaching experiments compared to proportions used in existing landfill risk assessments**

| | Leigh Marshes deionised water (%) | | | | Proportions used in existing risk assessments[1] (%) |
|---|---|---|---|---|---|
| Element | Min. | Max. | Range | Median | |
| Cd | 0.05404 | 0.14207 | 0.0880 | 0.08951 | 0.06-0.22 |
| Cu | 0.00124 | 0.00697 | 0.0057 | 0.00350 | 0.3-1.7 |
| Fe | 0.00003 | 0.00009 | 0.0001 | 0.00006 | 0.04-0.08 |
| Pb | 0.00003 | 0.00155 | 0.0015 | 0.00031 | 0.03-0.6 |
| Zn | 0.02408 | 0.09780 | 0.0737 | 0.08081 | 1.2-8 |
| [1]Source: Belevi and Baccini (1989) | | | | | |

**Table 4.39: Proportion of metals released from Hadleigh Marsh landfill matrix material during the leaching experiments compared to proportions used in existing landfill risk assessments**

| | Hadleigh Marsh deionised water (%) | | | | Proportions used in existing risk assessments[1] (%) |
|---|---|---|---|---|---|
| Element | Min. | Max. | Range | Median | |
| Cd | 0.07375 | 0.15454 | 0.0808 | 0.12726 | 0.06-0.22 |
| Cu | 0.02426 | 0.05364 | 0.0294 | 0.04025 | 0.3-1.7 |
| Fe | 0.00011 | 0.00027 | 0.0002 | 0.00014 | 0.04-0.08 |
| Pb | 0.00018 | 0.00086 | 0.0007 | 0.00034 | 0.03-0.6 |
| Zn | 0.02469 | 0.05683 | 0.0321 | 0.04553 | 1.2-8 |
| [1]Source: Belevi and Baccini (1989) | | | | | |

### 4.21.2 Variability within each landfill site

The proportions of metals released were highly variable (up to two orders of magnitude for some metals) between samples from the same landfill site leached in the same leachant type. For comparison, Belevi and Baccini's (1989) study only showed variability of one order of magnitude; however, they dried and ground their samples to homogenise them before leaching. The initial concentrations of the metals in the matrix material were also highly variable (up to one order of magnitude). Few correlations were found between the proportions of metals released and their initial concentrations in the matrix material. Where correlations did exist they were negative, suggesting that either as concentrations of the metals in the matrix material increased they sorbed to stronger binding sites (Hatje et al., 2003), metals in the higher concentration leachates reached saturation or that the matrix material with higher concentrations of metals contained insoluble metalliferous particles that are not released. The nature of the

association of metals with the waste would require further analysis to either characterise the waste material, e.g. using Scanning Electron Microscopy (SEM), or to explore the partitioning of metals to the waste, e.g. using sequential extractions, which is beyond the scope of this research.

### 4.21.3 Differences between landfill sites

The proportions of metals released in the same leachant type were significantly different between sites. The proportion of Zn released from the Leigh Marshes matrix material in deionised water was significantly higher than the proportion released from the Hadleigh Marsh matrix material. The proportions of Al, Cd, Fe, Mn and Ni released from the Hadleigh Marsh matrix material in deionised water were significantly higher than the proportions released from the Leigh Marshes matrix material.

The proportions of Fe and Zn released from the Leigh Marshes matrix material in artificial seawater were significantly higher than the proportions released from the Hadleigh Marsh matrix material. The proportions of Cd, Co, Ni and Pb released from the Hadleigh Marsh matrix material in artificial seawater were significantly higher than the proportions released from the Leigh Marshes matrix material.

No significant differences were found between the DO, redox potential, temperature or pH in the leachates for the two landfill sites in either deionised water or artificial seawater. For both leachant types the Hadleigh Marsh leachates had significantly higher DOC concentrations than the Leigh Marshes leachates. DOC can form complexes with metals increasing their mobility (Christensen et al., 1996). However, only negative correlations were found between DOC and the proportion of metals released in this research; therefore, the higher proportions of metals released from Hadleigh Marsh were not related to the higher DOC in the Hadleigh Marsh leachates.

The speciation of metals and their compounds within each site's matrix material are likely to have been different due to their different types and sources (Bódog et al., 1996). Metal speciation affects the strength of metal bonds to sediment and metal mobility (Ashraf et al., 2012). To determine the speciation of metals within the matrix material would have required analysis of the material using sequential extractions, which were beyond the scope of this research. In addition, as per the variations within

the individual sites discussed above, there may be differences in the content of insoluble metalliferous particles that were not sorbed to other materials within the two sites.

The difference in the volume of leachant added to the samples from each site also needs to be considered. The total volume of leachant, including the field moisture of the samples, is the same for both sites; however, the difference in added artificial seawater volume means the Hadleigh Marsh leachates contain marginally more added salts than the Leigh Marshes leachates. If the additional salt were responsible for the differences between the sites in the proportions of metals released then the proportions of all metals released from the Hadleigh Marsh samples would have been higher, which they were not (see section 4.21.4). Therefore, the difference in the volume of leachant added is not considered to have contributed significantly to the differences in metal proportions released between Hadleigh Marsh and Leigh Marshes.

### 4.21.4 Differences between freshwater and seawater

Significantly higher proportions of Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb and Zn were released from the matrix material from Leigh Marshes and Hadleigh Marsh in the artificial seawater leachates compared to the deionised water leachates. In addition, significantly higher proportions of Al were released from the Leigh Marshes matrix material in the artificial seawater leachates compared to the deionised water leachates. The greatest increase was 5,450% for Pb from the Leigh Marshes matrix material (based on median values).

The artificial seawater leachates contain significantly more DOC than the deionised water leachates, this is due to Na in the seawater competing for binding sites with Ca and Mg causing a loss of aggregate stability and release of organic matter (Green et al., 2008). DOC can form complexes with metals increasing their mobility (Christensen et al., 1996), but only negative correlations were found between DOC and the proportion of metals released in this research. Therefore, the higher proportions of metals released from waste in artificial seawater compared to waste in deionised water are not related to the higher DOC in the artificial seawater leachates.

The formation of soluble complexes of Al, Cd, Co, Cr, Fe, Mn, Ni and Zn with $Cl^-$ and $SO_4^{2-}$ anions, and competition for sorption sites between Cd, Cu, Pb and Zn, and $Ca^{2+}$, $K^+$, $Mg^{2+}$ and $Na^+$ cations can increase the mobility of metal contaminants in artificial

seawater (Cosovic et al., 1982; Sposito, 1996; Kraepiel et al., 1997; Zachara et al., 1987; Millero, 1998; Chapman and Wang, 2001; Acosta et al., 2011; Heiser et al., 2001) and can account for the increased proportions of Al, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb and Zn released in the artificial seawater leachates.

Aluminium in the Hadleigh Marsh samples was the only metal that did not significantly increase its released proportion in artificial seawater. No significant differences were found between the DO, redox potential, temperature or pH in the two leachant types for either of the two landfill sites. For both landfill sites, the artificial seawater leachates contain significantly more DOC than the deionised water leachates, but no correlation was found between the proportion of Al released and DOC. Therefore, it is likely that the reason for there being no significant difference in Al proportions released between the two leachant types is that Al speciation in the Hadleigh Marsh samples was such that it did not behave differently in the two leachant types.

### 4.21.5 Implications of differences in released proportions of metals between sites for landfill risk assessments

The results of this research show that the proportions of metals released by inundated landfill waste into solution are highly variable, differing by up to two orders of magnitude for some metals between samples of the same material in the same leachant type, and are significantly different between materials from different landfill sites. In addition, it has been found that the salinity of the water the waste is inundated with significantly changes the proportion of metals released. For Cd, Co, Cr, Fe, Mn, Ni, Pb and Zn the leachant salinity had a greater effect upon the proportion of metal release than the waste origin. However, for Al and Cu the waste source had a greater effect on the proportion of metal released than the leachant salinity.

Therefore, the results of this research should only be used to determine the effects of flooding and erosion on the sites sampled and should not be used to represent other sites even if they are of similar ages and contain similar materials. This is important when considering methods to determine which historic coastal landfills pose the greatest environmental risk, and suggests that risk ranking methods that assume that the proportions released are the same for all landfill sites (e.g. Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013) are potentially misleading in their

conclusions about which landfill sites pose the greatest pollution risk. This is supported by previous research into metal concentrations in leachates that concluded the behaviour of metals in leachates is complex and specific to individual landfill sites, and that results from one site should not be used to predict the behaviour of other sites (United States Environmental Protection Agency, 1992; Mukherjee et al., 2014).

## *4.22 Environmental impacts of released metals*

### 4.22.1 Assessment of the impact upon water quality by the release of metals from inundated, contained landfill waste using WAC

Hadleigh Marsh and Leigh Marshes landfills are both located on the Thames Estuary and are at risk of fluvial flooding from land drains and tidal flooding from the estuary. The flood risk for Leigh Marshes ranges from "very low" (>1 in 1000 return period) to "high" (<1 in 30 return period) (Environment Agency, 2016e). The flood risk for Hadleigh Marsh is not defined on Environment Agency flood maps, but the over topping return period is estimated at between 1 in 100 and 1 in 200 for the lowest point, and between 1 in 1,000 and 1 in 5,000 generally (Halcrow Group Ltd, 2012). Climate change effects will increase the flood risk for the sites as sea level rises and storm events become more frequent and intense, and for landfills further upstream, where the estuary is not yet fully saline, the salinity of floodwaters is also likely to increase. There is no evidence of hydrological connections between the waste in Leigh Marshes or Hadleigh Marsh and the estuary (Halcrow Group Ltd, 2012) suggesting that there has not previously been any saline water infiltrating into the sites. Therefore, it can be expected that tidal flooding of the sites would cause the first intrusion of saline water into their waste and metals currently bound to their waste will be released into leachates.

When considering the release of metals when a landfill site is inundated, but waste is not eroded, it is appropriate to use the WAC inert limit values to assess pollution risk as they were developed to assess leaching from fully encapsulated waste and, therefore, take into account attenuation of metals by the encapsulating materials. Under modern regulations, waste materials that generate leachates that exceed WAC inert limit values when tested according to BS EN 12457-2 are considered to pose a pollution risk even when leachate is being actively managed and they are not permitted to be landfilled within a groundwater source protection zone, on a major aquifer or below the water table where groundwater contributes significantly to river flow or other sensitive surface waters (Defra, 2010).

Figure 4.5 shows the comparison between the mass of metals released per kilogram of dry matrix material from the two sites in deionised water and artificial seawater to the

Waste Acceptance Criteria inert limit values, where available, i.e. Cd, Cr, Cu, Ni, Pb and Zn. Metal release per kg of dry matrix material in all of the deionised water leaches is below the WAC inert limit values, indicating that metals released from the matrix material pose no threat to water quality in a freshwater environment. However, the metals released per kg of dry matrix material during the artificial seawater leaches exceed the limit values for Cd and Zn in the majority of cases (i.e. median metal release > limit value). The limit value for Pb is only exceeded by a single outlier, which is an artificial seawater leach of the Leigh Marshes matrix material. No samples exceeded the limits for Cr or Cu.

Exceedance of the WAC inert limit values does not necessarily mean that leachates from the waste would be polluting in coastal waters. Although the limit values are used for assessing the pollution risk posed by leachates to all surface waters, they were originally developed for the protection of groundwater (Defra, 2010) and, therefore, do not account for differences between fresh, estuarine and coastal environments. The EQSs for the Protection of Surface Water permit higher concentrations of some metals in transitional and coastal waters compared to freshwaters before it is considered there will be an adverse impact upon water quality or aquatic life, e.g. Cr(III), and, depending upon freshwater hardness, Cd and Cu (Environment Agency, 2011; 2013c) (see Table 4.40). Iron, Ni and Pb have the same EQSs in fresh, transitional and coastal waters. Higher concentrations of Cr(VI), Zn and, depending upon freshwater hardness, Cd and Cu are permitted in the EQSs for freshwater than in the EQSs for transitional and coastal waters (Environment Agency, 2011; 2013c). Therefore, the use of the WAC inert limit values developed for freshwaters to assess historic coastal landfills creates the potential to underestimate the pollution risk posed to coastal waters by the leaching of Cd, Cr(VI), Cu and Zn, and overstate the risk posed by the leaching of Cr(III). This also suggests that the use of the WAC to test waste prior to disposal in present-day coastal landfills may underestimate the pollution risk posed to coastal waters by the leaching of Cd, Cr(VI), Cu and Zn. Therefore, there is a need for WAC limit values to be developed for coastal environments, not only for considering the impacts of historic landfills, but also to adequately assess whether waste materials being disposed of at present-day coastal landfills have the potential to pollute coastal environments.

If it is assumed that new coastal WAC limit values would differ from the existing WAC limit values in the same proportions as the EQSs for freshwater and coastal waters differ, then there would be no change in whether metal concentrations in the Hadleigh Marsh or Leigh Marshes leachates meet or exceed the limit values, i.e. the concentration of Cd and Zn in the artificial seawater leaches would still be of environmental concern. In actuality, whether the Cd and Zn concentrations would be detrimental to surface water quality would depend upon the capacity of the receiving waters to dilute their concentrations to below the limits set in the EQSs (Neuhold, 2013), see Section 4.22.2.

It should also be noted that, although the focus of this research is on surface water pollution, some historic coastal landfills are within groundwater source protection zones (SPZs), e.g. Temple Marsh in Kent is in SPZ2 and Broadness in Kent is in SPZ3. Only landfills with waste that produces leachates below the WAC inert limits when tested according to BS EN 12457-2 are permitted to be landfilled in those SPZs under current regulations. Sites that exceed the WAC inert limits are considered a pollution risk if located in those SPZs. The SPZs would be at risk of becoming polluted if metal mobilisation increased due to seawater flooding of the landfill sites; however, it is likely that the aquifers would no longer be used as water sources if saline intrusion occurred.

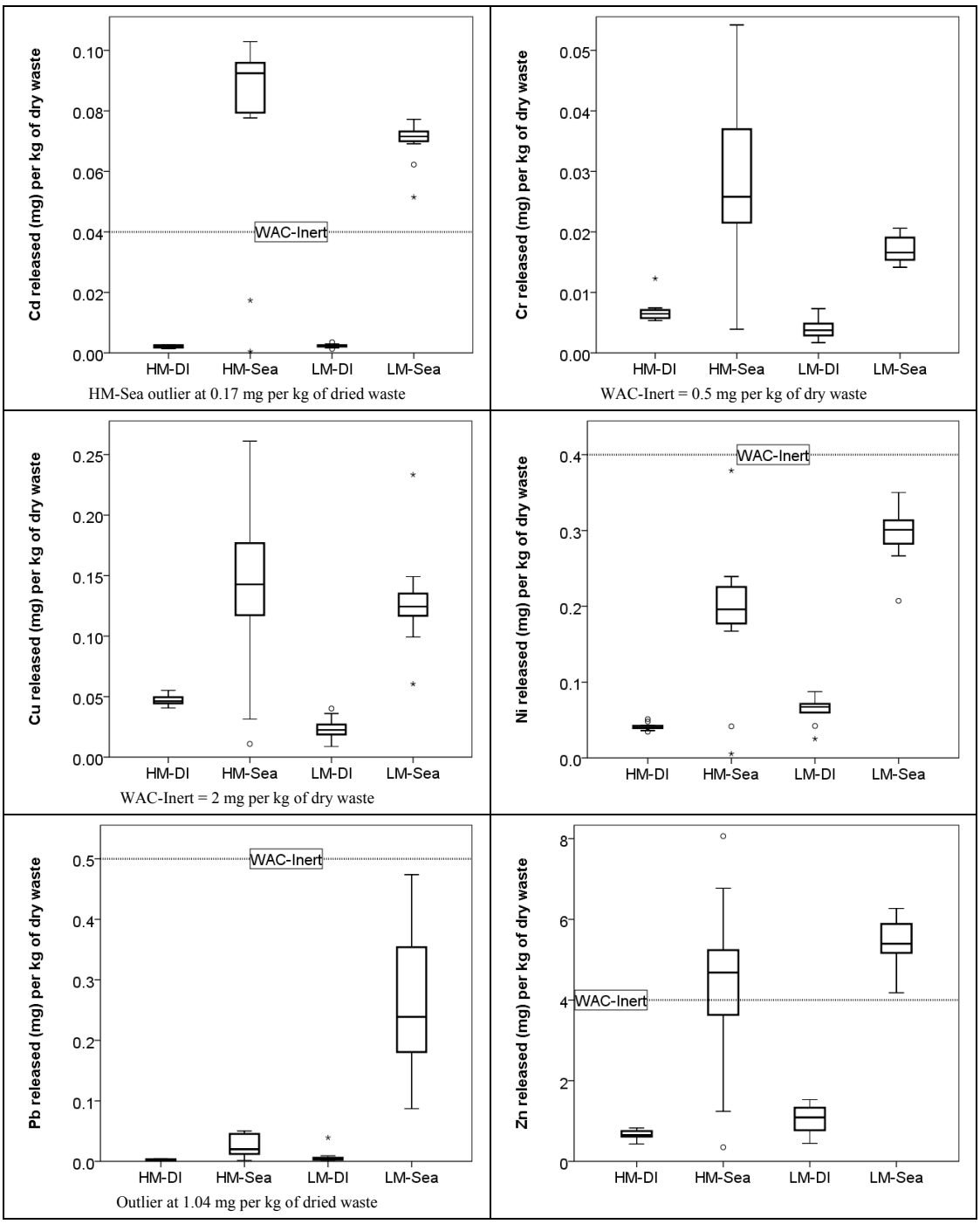



**Figure 4.5: Metal mass (mg) released per kg of dry waste (not blank corrected) compared to WAC limit values for landfilled waste (where available). HM=Hadleigh Marsh, LM=Leigh Marshes, DI=Deionised water leaches, Sea=Artificial seawater leaches**

**Table 4.40: Protection of Surface Water Quality EQSs in freshwater, transitional and coastal waters. Values are for dissolved concentrations, except where indicated (Environment Agency, 2011; 2013c)**

| | Inland surface waters (freshwater) EQS (µg $L^{-1}$) | | Other surface waters (transitional and coastal) EQS (µg $L^{-1}$) | |
|---|---|---|---|---|
| **Element** | **Annual average** | **Maximum allowable concentration** | **Annual average** | **Maximum allowable concentration** |
| Cd* | ≤0.08 (Class 1)<br>0.08 (Class 2)<br>0.09 (Class 3)<br>0.15 (Class 4)<br>0.25 (Class 5) | ≤0.45 (Class 1)<br>0.45 (Class 2)<br>0.6 (Class 3)<br>0.9 (Class 4)<br>1.5 (Class 5) | 0.2 | ≤0.45 (Class 1)<br>0.45 (Class 2)<br>0.6 (Class 3)<br>0.9 (Class 4)<br>1.5 (Class 5) |
| Cr(III) | 4.7 | 32 | n/a | n/a |
| Cr(VI) | 3.4 | n/a | 0.6 | 32 |
| Cu* | 1 (A)<br>6 (B)<br>10 (C)<br>28 (D) | n/a | 5 | n/a |
| Fe | 1000 | n/a | 1000 | n/a |
| Ni | 20 | n/a | 20 | n/a |
| Pb | 7.2 | n/a | 7.2 | n/a |
| Zn* | Total Zn<br>8 (A)<br>50 (B)<br>75 (C)<br>125 (D) | n/a | 40 | n/a |

*water hardness dependent
**Cd:** Class 1:<40mg $CaCO_3$ $L^{-1}$, Class 2: 40 to <50 mg $CaCO_3$ $L^{-1}$, Class 3: 50 to <100 mg $CaCO_3$ $L^{-1}$, Class 4: 100 to <200 mg $CaCO_3$ $L^{-1}$, Class 5 ≥200 mg $CaCO_3$ $L^{-1}$

**Cu and Zn:** A: 0-50mg $CaCO_3$ $L^{-1}$, B: 50-100 mg $CaCO_3$ $L^{-1}$, C: 100-250 mg $CaCO_3$ $L^{-1}$, D: 250+ mg $CaCO_3$ $L^{-1}$

### 4.22.2 Assessment of the impact upon surface water quality by the release of metals from eroded landfill waste using EQSs

There is a risk that the landfill sites will erode or catastrophically fail and release waste directly into the estuary. The life expectancy of Hadleigh Marsh embankment is not known, but it is recorded as having a high risk of failure due to its exceptionally weak foundations (Environment Agency, 1996), and Leigh Marshes' flood defences have an unmaintained life expectancy of 21-30 years (Environment Agency, 2010a). If the waste is inundated while the site is intact, e.g. by overtopping of the flood defence, it is likely that there will be significant attenuation of the metals within the waste itself, in its

encapsulating materials and in surrounding sediments (Christensen et al., 1994; Kjeldsen et al., 2002). However, if the waste were to be eroded and released onto the adjacent saltmarsh or into surface waters it is likely that any metals released to solution would directly enter the water column. Although the proportions of metals that were released in the leaching experiments were very small, it is possible that the total mass of metals released would be environmentally significant as there are very large volumes of waste materials within the sites. Whether the released metals would pose a pollution risk in this scenario would depend upon the rate of waste erosion and the capacity of the receiving water to dilute the metals as they are released.

As there is currently no way to determine the rate of erosion of these sites, the worst case scenarios of the entirety of Leigh Marshes or a single waste cell of Hadleigh Marsh rapidly failing were assessed. The total mass of Cu, Ni and Zn that could be released from a single Hadleigh Marsh landfill waste cell in river or seawater is less than the daily output of most sewage treatment works (STWs), and the Thames Estuary has at least eleven STWs discharging into it (Stevenson and Ng, 1999). The total mass of Cu and Ni that could be released from Leigh Marshes landfill in river water is less than the combined daily output of the two largest sewage treatment works (STW). The total mass of Zn in river water and Cu, Ni and Zn in seawater that could be released from Leigh Marshes landfill is 84%, 48%, 116% and 418% of the average daily total input of those metals from all known sources respectively. However, it should be remembered that this represents the worst case scenario and it is unlikely that the entirety of Leigh Marshes landfill would erode in a single day as the site is protected by a flood defence and extends 200-350 metres inland.

In addition, the Thames Estuary is macrotidal and has a tidal prism volume of 585,000,000 $m^3$ (Mikhailov and Mikhailova, 2012), if it is assumed that the metals released would be diluted by a minimum of one tidal prism volume, i.e. erosion of the material and release of the metals to solution takes at least 12 hours, then the maximum concentrations of metals reached for Leigh Marshes would be at least one order of magnitude below the EQSs for the Protection of Surface Water annual average and maximum allowable concentrations, and for Hadleigh Marsh they would be at least two orders of magnitude below the annual average and maximum allowable concentrations (Table 4.41 and Table 4.42). This is actually a significant underestimate of the dilution

that would take place as it does not account for the water present in the estuary that is not part of the tidal prism volume. In addition, breaching of a site is likely to occur during extreme events such as flooding or tidal surges when the dilution will be even greater than normal. Therefore, metals released from waste eroded from these sites are unlikely to have a significant impact upon water quality in the Thames Estuary and would not impact upon the River Thames's status under the Water Framework Directive (WFD), as the WFD surface water classification system only considers annual average concentrations.

**Table 4.41: Maximum dissolved metal concentrations in water resulting from the erosion of Leigh Marshes landfill in a single tidal cycle, based on a tidal prism volume of 585,000,000 $m^3$, compared to surface water quality EQSs**

| Element | EQS annual averages (ng $L^{-1}$) | | EQS Max allowable concentrations (ng $L^{-1}$) | | Max. concentration when diluted by tidal prism volume (ng $L^{-1}$) | |
|---|---|---|---|---|---|---|
| | River | Sea | River | Sea | Eroded into river water | Eroded into seawater |
| Al | n/a | n/a | n/a | n/a | 41.7 | 106 |
| Cd | 80[2] | 200 | 450[2] | 450[2] | 1.16 | 33.4 |
| Co[1] | 3,000 | 3,000 | 100000 | 100,000 | 2.11 | 4.98 |
| Cr III/VI | 4,700/3,400 | (n/a)/600 | 32,000/(n/a) | (n/a)/32,000 | 1.75[3] | 7.73[3] |
| Cu | 1,000[2] | 5,000 | n/a | n/a | 10.5 | 58.0 |
| Fe | 1,000,000 | 1,000,000 | n/a | n/a | 23.4 | 93.6 |
| Mn[1] | 30,000 | n/a | 300,000 | n/a | 0.63 | 26.8 |
| Ni | 20,000 | 20,000 | n/a | n/a | 31.4 | 140 |
| Pb | 7,200 | 7,200 | n/a | n/a | 1.90 | 111 |
| Zn | 8,000[2] | 40,000 | n/a | n/a | 507 | 2,513 |

[1]Protection of Aquatic Life EQS. [2]water hardness dependant, lowest limit shown. [3]$Cr_{total}$.

**Table 4.42: Maximum dissolved metal concentrations in water resulting from the erosion of a single Hadleigh Marsh waste cell in a single tidal cycle, based on a tidal prism volume of 585,000,000 $m^3$, compared to surface water quality EQSs**

| Element | EQS annual averages (ng $L^{-1}$) | | EQS Max allowable concentrations (ng $L^{-1}$) | | Max. concentration when diluted by tidal prism volume (ng $L^{-1}$) | |
|---|---|---|---|---|---|---|
| | River | Sea | River | Sea | Eroded into river water | Eroded into seawater |
| Al | n/a | n/a | n/a | n/a | 4.13 | 2.79 |
| Cd | 80[2] | 200 | 450[2] | 450[2] | 0.04 | 1.72 |
| Co[1] | 3,000 | 3,000 | 100000 | 100,000 | 0.03 | 0.08 |
| Cr III/VI | 4,700/3,400 | (n/a)/600 | 32,000/(n/a) | (n/a)/32,000 | 0.12[3] | 0.48[3] |
| Cu | 1,000[2] | 5,000 | n/a | n/a | 0.86 | 2.66 |
| Fe | 1,000,000 | 1,000,000 | n/a | n/a | 1.04 | 4.88 |
| Mn[1] | 30,000 | n/a | 300,000 | n/a | 0.03 | 0.60 |
| Ni | 20,000 | 20,000 | n/a | n/a | 0.74 | 3.65 |
| Pb | 7,200 | 7,200 | n/a | n/a | 0.04 | 0.37 |
| Zn | 8,000[2] | 40,000 | n/a | n/a | 12.1 | 87.2 |

[1]Protection of Aquatic Life EQS. [2]water hardness dependant, lowest limit shown. [3]$Cr_{total}$.

## *Summary*

This research has found that the proportion of each metal released during leaching varies greatly between samples from a single landfill site and between landfill sites. This is probably due to variations in initial metal concentrations and speciation in the waste, and the presence of insoluble metalliferous particles in variable quantities. These differences mean that the proportions of metals released from material from one landfill site cannot be used to predict behaviour at other similar landfill sites, suggesting existing risk assessments that use standardised proportions to represent all sites are misleading in their conclusions and an alternative approach is needed.

It has also been found that the proportions of metals released are significantly increased if the leachant is saline, which is probably due to the formation of soluble complexes between metals and saltwater anions, and competition for sorption sites between metals and saltwater cations. This increase suggests that when climate change effects lead to increased saline intrusion into estuaries there will be increased metal leaching from historic coastal landfill sites that are hydrologically connected to estuarine waters. Current European regulations and British Standard methods for assessing leaching from landfill waste do not assess the effect of saltwater ingress into landfill sites. There is a clear need for them to be updated to consider the effects of saltwater intrusion when classifying waste for disposal at present-day coastal landfill sites, because although these sites have liners and leachate management systems it is known that leachate will escape even from well-engineered, lined landfills (Arneth et al., 1989; Bagchi, 1994; Allen, 2001).

Metals released from the historic coastal landfills analysed for this research, Leigh Marshes and Hadleigh Marsh, are unlikely to pose a risk to surface water quality due to the low proportions of metals mobilised (typically «1%) and due to high levels of dilution in the Thames Estuary. Total metal loads and the proportions of metals released can vary significantly between landfill sites, but it is also unlikely that metals released from other historic coastal landfills due to inundation will pose a pollution risk even for larger sites in lower flow locations due to attenuation of metals within the waste itself and in its capping materials, and the high levels of dilution that occurs as a result of river flow and tidal exchange in tidal environments. However, it cannot be assumed that metals released by waste eroded from the other 1262 historic coastal landfills would not

pose a risk to surface water quality as eroded waste would release metals directly into surface waters and the volume of waste and the dilution needs to be assessed for each site individually. In addition, only the pollution risk from a limited suite of metals has been considered and the effect of other landfill contaminants, e.g. Ammonium-N, on surface water quality warrant further investigation.

# Chapter 5. Developing a risk screening assessment for ranking historic coastal landfills by pollution risk

## *5.1 Introduction, aims and objectives*

The results of the previous chapters show that inundation of historic coastal landfills is unlikely to have a significant effect upon the surrounding environment if the waste is contained, but there is the potential for pollution to occur if historic coastal landfills are breached and waste is eroded into the intertidal zone and/or tidal waters. However, simply knowing that a site may be polluting if it erodes is not sufficient when determining management strategies. An understanding of the pollution risk is also required and, when allocating resources for the management of multiple sites, an understanding of which sites pose the greatest pollution risk is also necessary. Therefore, this chapter is focused on the development of a new risk screening assessment method for historic coastal landfill sites.

Risk is typically considered as a function of the probability of something happening and the consequences of it happening (Wamsley, 2015). There are many factors that may influence the probability of the release of contaminated materials from historic coastal landfill sites, including the condition of any flood defences, the design standard of those defences and the probability of the design standard being exceeded, whether the coast is eroding at its location, site exposure, whether the site's defences are vegetated or armoured, the presence of any buffers, e.g. saltmarshes, site maintenance regime, climate change related sea-level rise and storm severity. The consequences of the pollution scenarios occurring are dependent on the vulnerability of the receptors (Wamsley, 2015).

The vulnerability of the receptors can be considered as the probability that the receptors will be affected by hazards, also known as threats, stressors or drivers, which for chemical hazards is often considered in terms of a dose-response relationship (Gormley et al., 2011). Therefore, the consequences of contaminated materials being released will depend upon the quantity of materials released and their contaminant loads, contaminant bioavailability and mobility, dilution by the receiving waters, and receptor sensitivity to those contaminants. In turn, the quantity of materials released will depend on many of the same factors as the probability of contaminated material release, plus the size of the

landfill, whether it is divided into structurally stable cells, the mechanical properties of the waste, e.g. waste cohesion, the shape of the landfill, i.e. the proportion of it adjacent to the coast, and how quickly any breach can be repaired.

Combining diverse data types, representing complex and interacting parameters, into a readily understood form that indicates their combined effect can be achieved using index and indicator methods (Ramieri et al., 2011). However, many of these data will not be readily available and would require impracticable levels of resources to obtain for all of the 1264 historic coastal landfills around England. Coastal defence budgets are already stretched and central government funding is continually being reduced whilst flood defence maintenance costs are rising by £10-£30 million plus inflation per year (Bennett and Nartwell-Naguib, 2014). Where detailed data are not readily available to assess risk at local, regional or national scales Rosendahl Appelquist and Balstrøm (2014) propose a three step approach to assessment, where steps 1 and 2 are used for regional or national scale assessments and step 3 is only used for local scale assessments:

Step 1. High level initial screening using remote sensing and existing data to gain a cost-efficient, relatively low accuracy overview of the risk.
Step 2. Field verification of the data used in step 1.
Step 3. Systematic and detailed field investigations for high accuracy, local level assessments of risk hot-spots identified in steps 1 and 2.

This approach has the advantage of reducing expenditure on site investigations and providing a method to prioritise resources when there are multiple sites to manage. It has the disadvantage that existing data may not highlight factors that increase risk, e.g. records may not show that a site has already started to erode.

There are a number of existing screening index and indicator risk assessment methods for landfills which were developed in other countries for use in specific environments. Only two consider the erosion of waste as a pollutant pathway: first, the method used by the Alaskan Waste Erosion Assessment and Review (WEAR) project (Alaska Department of Environmental Conservation, 2015) requires site visits to obtain the necessary assessment data, and second, the Austrian method (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013), which is designed for fluvial

environments and only considers erosion during flood events, would require new flood models for each site. Hence both methods are impracticable for ranking all 1264 of England's historic coastal landfills due to the resources required and a new region specific method is necessary.

Therefore, the aim of this chapter is to develop a high-level risk screening assessment methodology that focusses on the risk to the intertidal zone and tidal waters from eroding historic coastal landfills. The specific objectives are to:

1. Create a risk screening assessment for ranking the relative pollution risk of historic coastal landfill sites that combines the following:
    a. Assessment of the risk of waste being released - identify suitable parameters and existing datasets that represent the potential for historic coastal landfill sites to erode.
    b. Assessment of the risk to receptors from eroded waste - identify suitable parameters and existing datasets that represent the risk to the environment if historic coastal landfill sites erode and waste is released to the coastal zone.
2. Test the proposed screening assessment methodology on eight landfills in Essex to rank them by relative pollution risk.

## 5.2 Approach to developing a screening assessment

The assessment of the risk of pollution in the intertidal zone and/or tidal waters from eroding historic coastal landfills can be considered in two stages: the first stage is to assess the risk of waste being released, and the second is to assess the risk to various receptors from the released waste. Although the risk of historic coastal landfills eroding is not well studied, there are a wide variety of coastal vulnerability assessments that attempt to rank the vulnerability of human infrastructure in general to coastal processes and climate change effects (e.g. McLaughlin and Cooper, 2010; Ramieri et al., 2011). These have the potential to be adapted and optimised for assessing the risk of landfills releasing waste. Similarly, there are a number of existing screening index and indicator risk assessment methods for landfills that have the potential to be adapted and optimised for assessing the risk to various receptors from the released waste (e.g. Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013; Alaska Department of Environmental Conservation, 2015).

Both coastal vulnerability and landfill screening assessments provide relative, not absolute, indications of risk and typically assess the risk by considering the vulnerability of receptors to specific hazards. In coastal vulnerability assessments the hazards are internal or external risk factors or situations that may adversely affect a receptor in a system (Sayers et al., 2003; Kumar et al., 2010; Wamsley, 2015), whereas in coastal landfill risk assessments a hazard is specifically defined as "a substance in or under the land that has potential to be hazardous to human health or the environment" (Cooper et al., 2013, p. xvi). In this research, in the first stage of the assessment historic coastal landfills are the receptors, but in the second stage they are the source of the hazard and the receptors are then flora, fauna, protected sites, etc. in the intertidal zone and/or tidal waters. The relationship between hazards/drivers, vulnerability and overall risk is shown in Figure 5.1.

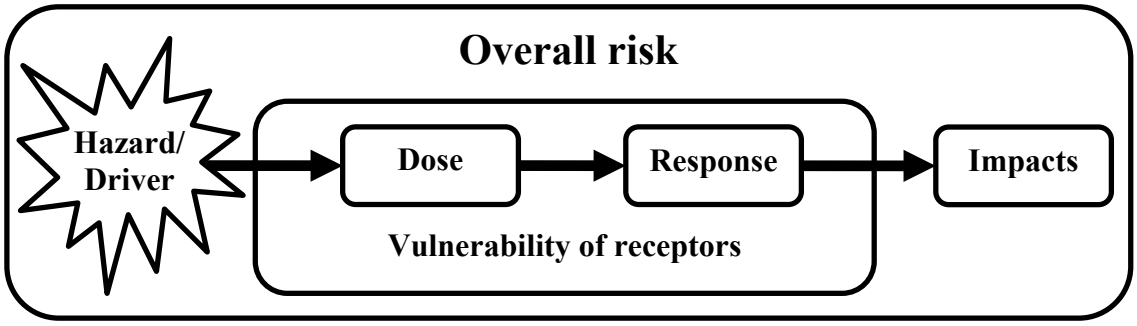



**Figure 5.1: The relationship between overall risk, hazard/driver and vulnerability (adapted from Gormley et al., 2011; Wamsley, 2015)**

There exist general guidelines for creating risk screening assessments. Assessments should be kept simple and use the best available data that either measure the risk factors directly or act as a proxy (Rygel et al., 2006; Wamsley, 2015). As few parameters as possible should be used to represent the main driving factors of the risk scenarios being considered, and data availability and the assessment scale should drive their selection (McLaughlin and Cooper, 2010; Wamsley, 2015). There can be strong correlations between parameters, which means increasing the number used will not necessarily increase the assessment method's accuracy (McLaughlin and Cooper, 2010) and restricting the number of parameters, which reduces the time and cost of the assessment, increases the usability of methods and the likelihood of them being adopted (Singh et al., 2009). Parameters are often assigned relative severity scores when used in screening/ranking assessments (Ramieri et al., 2011). For quantitative data, e.g. wave height, the severity scores are typically based on numerical ranges, for qualitative data, e.g. landform resistance to erosion, value judgements may be used to assign severity scores (Ramieri et al., 2011; Wamsley, 2015). Therefore, parameters with quantitative datasets and with qualitative datasets can both be used in the assessments, providing relative severity scores can be applied to them (Singh et al., 2009).

Two main reporting approaches are used in existing coastal vulnerability and landfill risk ranking research: index methods and indicator methods. Index methods are commonly used for global or regional scale assessments to simplify diverse data types, representing complex and interacting parameters, into a readily understood form that indicates their combined effect (McLaughlin and Cooper, 2010). Index methods usually combine the severity scores from individual parameters into a single, often unitless, number that does not explain the assumptions and aggregations that led to it unless the

calculation method is also given (Ramieri et al., 2011). In contrast, indictor methods express their findings as a series of independent factors e.g. pressures, sensitivity and damage, each represented by groups of indicators (parameters). The individual indicators may also be combined into a final summary indicator. The indicator method makes it easier to understand the critical factors in the published results, but, unless a summary indictor is included, ranking by risk is easier using an index method (Ramieri et al., 2011). There is not always a clear distinction between the two methods as in addition to the overall index value some index methods report sub-indices, but in less detail than the indicator methods (McLaughlin and Cooper, 2010; Ramieri et al., 2011).

This research uses an index approach, first considering the risk of waste release and the environmental risk as separate indices, hereafter referred to as the waste release index and pollution index respectively, and then combining them into an overall risk index. To determine these indices, four sub-indices have been created: coastal drivers (e.g. wave action), landfill vulnerability (i.e. likelihood of the landfill releasing waste), landfill hazard (representing volumes and toxicity of waste released) and environmental vulnerability (i.e. likelihood of environmental harm from the released waste). The relationship between the sub-indices, indices and overall risk index is illustrated in Figure 5.2. The index approach was chosen to inform the end-user of how the overall risk was determined without overwhelming them with data by reporting every parameter considered. The method used in this research has been designed such that the higher the index value the greater the risk and hence the higher the priority of the historic coastal landfill for further investigation or remedial works. Increasing index values corresponding to increasing risk is consistent with the approach of many coastal vulnerability (e.g. Palmer et al., 2011; Gill et al., 2014; Denner et al., 2015) and landfill risk ranking assessments (e.g. Sharma et al., 2008; Alaska Department of Environmental Conservation, 2015). The severity scoring systems used for ranking parameters are commonly on a four point, (e.g. Palmer et al., 2011; Khouakhi et al., 2013; Denner et al., 2015) or five point scale (e.g. McLaughlin and Cooper, 2010; Gill et al., 2014; Musekiwa et al., 2015). This research uses a five point severity scale for each parameter. Wherever possible severity scores from existing risk assessment methods have been utilised, but new severity scoring systems are proposed for parameters where necessary. All of the severity scores are provisional and it is intended that the scores are reviewed with appropriate experts and regulators before a national

scale assessment is undertaken. The process of surveying and consulting with experts on coastal processes, landfill engineering stability and contamination, and ecology that would be necessary to achieve consensus on appropriate scores was beyond the scope of this research.

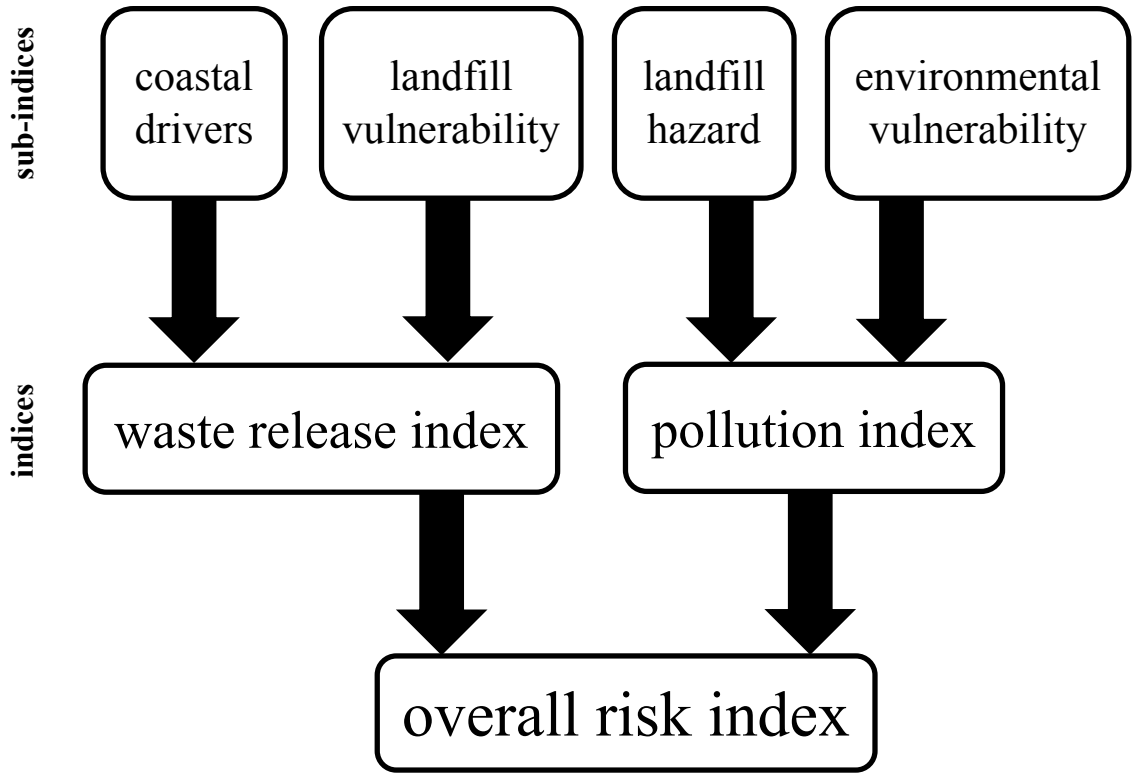



**Figure 5.2: Flow chart showing the relationship between the sub-indices, indices and overall risk**

There are many approaches to combining severity scores to give values for the sub-indices and overall risk index, these include simple summing, weighted summing, using the mean, modified product mean or square root of the product mean (Ramieri et al., 2011). Summation methods are less sensitive to variations of individual severity scores than product methods whilst still being able to differentiate indices when variations occur in several parameters (Gornitz and White, 1992; McLaughlin and Cooper, 2010; Ramieri et al., 2011). Hence, summation methods either with (e.g. Sharma et al., 2008; Palmer et al., 2011; Denner et al., 2015) or without (e.g. Khouakhi et al., 2013; Musekiwa et al., 2015) weightings are the most commonly used approach in the coastal vulnerability and landfill risk ranking methods reviewed in this research.

The WEAR method (Alaska Department of Environmental Conservation, 2015) used a variation on the weighted method where two sub-indices (contaminant risk and erosion

risk) were reported, but they were not combined to give an overall risk index value. Instead sites that were in the top 25% of scores for both sub-indices were considered to pose the greatest risk, i.e. were risk hotspots, this has the obvious disadvantage that if additional sites are added to the assessment the sites that were considered hotspots may change.

Methods that have used weighted summations have determined the weightings by carrying out extensive surveys of experts to obtain a consensus opinion as to which parameters are the most important to the assessment being carried out (e.g. Kumar and Alappat, 2005; Sharma et al., 2008; Singh et al., 2009). To date there have been a very limited number of coastal landfill breaches, and there have been few studies of the effects of leachates on estuarine and marine environments and none on the effects of landfill debris (Pope et al., 2011; Njue, 2012). Therefore, data do not exist that allow weighting of the parameters to be included in the assessment method based on existing literature. The process of surveying and consulting with experts on coastal processes, landfill engineering stability and contamination, and ecology that would be necessary to achieve consensus on appropriate weightings was beyond the scope of this research. Consequently, this research uses a summation method without weightings being applied to the severity scores to determine the values of the sub-indices (Equation 5.1). Where a different number of parameters are used for each of the sub-indices, normalising each sub-index value to a percentage allows them to be combined into the overall risk index without any one sub-index dominating the overall risk score (McLaughlin and Cooper, 2010). Therefore, the four sub-indices were normalised to percentages using Equation 5.2 before being combined into the waste release index and pollution index using Equation 5.3 and Equation 5.4 respectively. The overall risk index was then calculated using Equation 5.5. All three indices have value ranges from 0 to 100.

**Equation 5.1: Calculation of the sub-indices**

$$Sub\text{-}index = \sum severity\ scores$$

**Equation 5.2: Calculation of the normalised sub-indices (after McLaughlin and Cooper, 2010)**

$$Normalised\ sub\text{-}index = \frac{\sum severity\ scores - min.possible\ score}{max.possible\ score - min.possible\ score}\ x\ 100$$

**Equation 5.3: Calculation of the waste release index**

$$Waste\ release\ index = \frac{normalised\ coastal\ drivers + normalised\ landfill\ vulnerability}{2}$$

**Equation 5.4: Calculation of the pollution index**

$$Pollution\ index = \frac{normalised\ landfill\ hazard + normalised\ environmental\ vulnerability}{2}$$

**Equation 5.5: Calculation of the overall risk index**

$$Overall\ risk\ index = \frac{waste\ release\ index + pollution\ index}{2}$$

## 5.3 Selection of parameters to determine the coastal drivers and landfill vulnerability sub-indices

### 5.3.1 Parameters used in existing coastal vulnerability assessments

The coastal driver and vulnerability related parameters used in a selection of coastal vulnerability assessment methods that include erosion as one of the risks are shown in Table 5.1. Similar to landfill risk assessments, these methods have usually been developed for assessment of a single geographical region and use existing data and a scoring system uniquely tailored for that region. The Coastal Hazard Wheel (CHW) method is an exception, it aims to be applicable for regional and national planning for all coastal environments and considers gradual inundation, flooding, saltwater intrusion, ecosystem disruption and erosion hazards (Rosendahl Appelquist, 2013). The main factors highlighted in the CHW method as important for assessing coastal erosion are the geomorphological type of the coast line, coastal slope, wave exposure, sediment balance, storm climate, tidal range and vegetated areas. The CHW considers storm climate in terms of the presence/absence of cyclone activity, but there exists little variation in mean wind speeds and wind gust speeds around England (Met Office, 2016) and, therefore, storm climate was not considered in this research.

Only the multi-scale coastal vulnerability index for Northern Ireland (NI-CVI) (McLaughlin and Cooper, 2010) and coastal vulnerability and environmental risk assessment of Loughor Estuary in Wales (Denner et al., 2015) use methods specifically developed for UK coastal environments. Data available for England are indicated in Table 5.1 and explored in more detail in section 5.3.3. As this research was interested in erosion of historic coastal landfills, not of the natural coastline, features of the landfills were considered in place of the natural geomorphology and underlying geology.

The British Geological Survey (BGS) is currently developing a bespoke Coastal Vulnerability Index (CVI) for Great Britain, which will provide a qualitative risk rating for coastal erosion and flooding derived from quantitative data (G. Jenkins, BGS, pers. comm., email, 13/5/2016). The presentation format of the CVI's outputs is currently being assessed and it was expected to be released in July 2016 as a licensable product (G. Jenkins, BGS, pers. comm., email, 13/5/2016), but has not been released at the time of writing (October 2016). The CVI may be suitable for integration into a landfill

screening method in the future, depending on the ease of integrating its outputs with landfill and ecological location datasets.

**Table 5.1: Parameters used in coastal vulnerability assessments to represent coastal drivers (coastal processes and climate change effects) and the vulnerability of receptors**

| Coastal driver related parameters | PCVI | Moroccan | S. African | CHW | NI-CVI* | KwaZulu | Loughor | Existing datasets available for sites in England? |
|---|---|---|---|---|---|---|---|---|
| Relative sea level change (includes ground movement, e.g. isostatic adjustment) | ✓ | | ✓ | | | | | Some |
| Mean tidal range | ✓ | | ✓ | ✓ | ✓ | | | Yes-but not free |
| Mean wave height | ✓ | | ✓ | ✓ | | | | Some |
| Wave orientation to shore | | ✓ | | | | | | Some |
| Wave energy (based on fetch) | | | | ✓ | | | | Some |
| Wave energy (based on prevailing winds) | | | | | ✓ | | | Some |
| Maximum wave height | | | ✓ | | ✓ | | | Some |
| Anthropogenic impacts, e.g. dredging | | | ✓ | | | | | No |
| Inundation/flooding | | | | ✓ | | | | Yes |
| Storm climate (cyclone region) | | | | ✓ | | | | Yes |
| **Vulnerability related parameters** | | | | | | | | |
| Coastal geomorphology, e.g. cliffs, beaches | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | Some |
| Underlying geology | | | ✓ | | ✓ | | | Yes |
| Coastal elevation | | | ✓ | | ✓ | | | Yes |
| Shoreline erosion/sediment balance | ✓ | | | ✓ | | | | Some |
| Coastal slope | ✓ | ✓ | ✓ | ✓ | | | ✓ | No |
| Distance to 20 m isobaths | | | | | | ✓ | | Yes-but not free |
| Beach width | | ✓ | ✓ | | | ✓ | ✓ | Yes-foreshore |
| Dune width | | | | | | ✓ | ✓ | No |
| Vegetated buffer zones, e.g. saltmarshes | | ✓ | | ✓ | | ✓ | ✓ | Some types |
| Anthropogenic impacts, e.g. flood defences | | | ✓ | | | | | Yes |
| Land use | | | | | ✓ | | | Yes (all landfills) |

PCVI-Physical coastal vulnerability index Malaysia (Gill et al., 2014)
Moroccan-Vulnerability assessment of Al Hoceima bay (Khouakhi et al., 2013)
S. African-An assessment of coastal vulnerability for the South African coast (Musekiwa et al., 2015)
CHW-Coastal hazard wheel for multi-hazard assessment (Rosendahl Appelquist and Halsnæs, 2015)
NI-CVI- Multi-scale coastal vulnerability index for Northern Ireland (McLaughlin and Cooper, 2010)
*only parameters used for the national scale assessment are shown here
KwaZulu-Preliminary coastal vulnerability assessment for KwaZulu-Natal, SA (Palmer et al., 2011)
Loughor-Coastal vulnerability and environmental risk, Loughor Estuary, Wales (Denner et al., 2015)

### 5.3.2 Parameters used in existing landfill screening methods to assess erosion

The driver and vulnerability related parameters used in existing landfill screening methods to assess landfill erosion are shown in Table 5.2. The WEAR method (Alaska Department of Environmental Conservation, 2015) was a long-term investigation (4 years) of a large number of sites (716) and included site visits to obtain data not readily available in desktop studies (marked in Table 5.2 as not available for England). The Austrian method (Neuhold and Nachtnebel, 2011; Neuhold, 2013) used site specific modelling to assess a limited number of landfills in detail and only considered erosion during flood events, whereas the WEAR method considered continual erosion. Data available for England are indicated in Table 5.2 and explored in more detail in section 5.3.3.

In addition to data used in landfill screening/ranking methods, CIRIA C718 'Guidance on the management of landfill sites and land contamination on eroding or low-lying coastlines' (Cooper et al., 2013) suggests the inclusion of data relating to the condition of flood defences (where present) and the probability of them failing. Availability of such data are also explored in section 5.3.3

**Table 5.2: Vulnerability related parameters used in existing landfill screening methods that consider erosion**

| Pathway parameters | Method | | Existing datasets available for sites in England? |
|---|---|---|---|
| | Austrian | WEAR | |
| Years until erosion (calculated = distance to erosion/rate of erosion) | | ✓ | Erosion rates for some locations |
| Erosion type, e.g. seepage, wave action, precipitation | | ✓ | No |
| Key erosion factors, e.g. tides, storm surges, flooding, human influences | | ✓ | No |
| Erosion symptoms, e.g. undercuts, root exposure, waste exposure | | ✓ | No |
| Soil class, e.g. clay, silts, organic – resistance to erosion | | ✓ | No* |
| Mitigation, e.g. attempts to control erosion, presence of flood defences | | ✓ | Yes |
| Topography | ✓ | | Yes |
| Surface water and drainage, including flood probability | ✓ | | Yes |
| Floodwater velocity and shear stress | ✓ | | Pluvial flooding velocity only |
| Flood depth | ✓ | | Pluvial flooding only |
| Distance to water course | | ✓ | Yes |

Austrian = Austrian fluvial flooding of landfills risk assessment (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013)

WEAR = Waste Erosion Assessment and Review project (Alaska Department of Environmental Conservation, 2015)

*Soil class maps exist for England (e.g. British Geological Survey, 2016c), but they do not show soils for made ground, i.e. they do not show the soil class of capping materials at landfill sites.

### 5.3.3 Coastal drivers (of erosion) related data available for England

The main factors to be considered as coastal drivers of erosion in the UK environment are wave energy, tidal range and flooding (see sections 5.3.1 and 5.3.2). The best available data that either measure these directly or are suitable proxies are considered below. The severity scores assigned for the wave energy, tidal classification and flooding parameters are new for this research and are provisional pending future refinement of the risk screening assessment through consultation with relevant experts.

#### 5.3.3.1 Wave energy

Wave energy hitting the shoreline depends upon the height of waves, their orientation to shore, wave fetch, and width and vegetation of any buffer zones (Möller and Spencer, 2002; Defra and Environment Agency, 2007a; Rosendahl Appelquist, 2013). Currently, there are limited data available for most of these factors in England, the BODC website (British Oceanography Data Centre, 2016) holds some wave height data, but it is of low resolution and, therefore, not suitable for assessing short sections of coastline, which would be necessary to rate individual landfill sites. There are twenty-two Shoreline Management Plans (SMPs), covering the entire coast of England and Wales (Environment Agency, 2009a). The SMPs contain some information about wave orientations to shore and wave fetch, but the available data are inconsistent and the SMPs do not include the entire tidal extent of estuaries and rivers, e.g. the Thames Estuary is covered by Thames Estuary 2100 not a SMP (Environment Agency, 2012b). Therefore, the SMP data are not suitable for use in a national scale assessment. In the absence of wave data, free fetch can be used to classify coasts as protected (waterbody width <10 km), moderately exposed (10 km< waterbody width <100 km) or exposed (100 km< waterbody width) (Rosendahl Appelquist, 2013). Coastlines with a free fetch greater than 10 km may also be classed as protected if the local geology or wind and wave climate is such that wave action is limited, this is indicated by the presence of saltmarshes (Mangor, 2004; Rosendahl Appelquist, 2013). Free fetch can easily be determined using most maps and a GIS dataset of saltmarsh extents is available for EA partners to download for free from data.gov.uk (UK Government, 2016), therefore, this categorisation approach, i.e. rating sites as protected, moderately exposed or exposed, was used for ranking the relative impact of waves on historic coastal landfill sites. The severity score associated with wave energy was assigned using the decision tree shown in Figure 5.3.

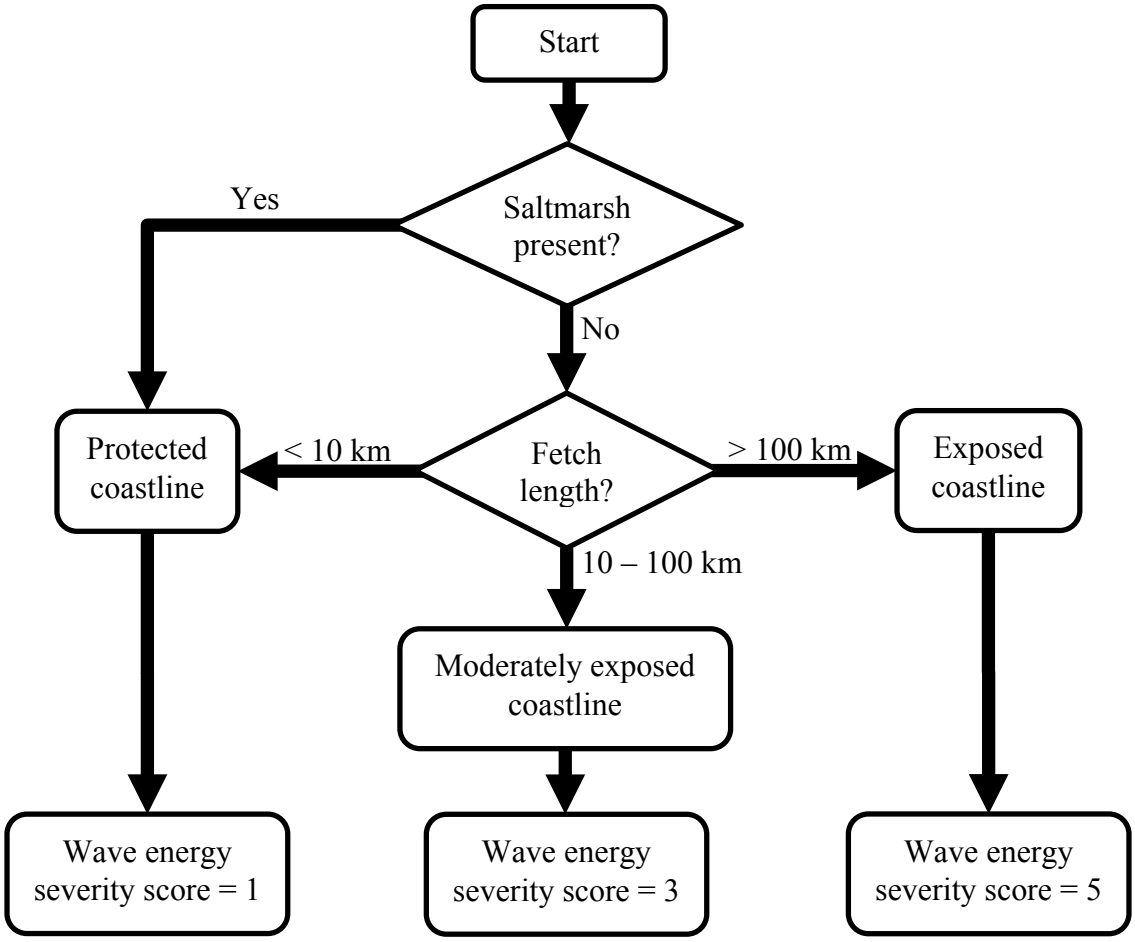



**Figure 5.3: Wave energy severity score determination (categorisation after Mangor, 2004; Rosendahl Appelquist, 2013)**

#### 5.3.3.2 Tidal classification

The tidal range influences how vulnerable coastlines are to wave energy (McLaughlin and Cooper, 2010) and flooding (Rosendahl Appelquist, 2013). The greater the tidal range the lower the probability that high tide and high waves will coincide, hence the probability of wave related erosion (McLaughlin and Cooper, 2010) and the probability of flooding (Rosendahl Appelquist, 2013) are reduced. In addition, wide intertidal zones in which wave energy can dissipate are often present in areas with high tidal ranges (McLaughlin and Cooper, 2010). The tidal classification, i.e. whether it is macrotidal (> 4 m range), mesotidal (2-4 m range) or microtidal (<2 m range), is considered adequate to assess tidal range as a hazard (Davies and Moses, 1964; Rosendahl Appelquist, 2013) and can be found for all British estuaries in a free to access JNCC report (Davidson, 1991). Therefore, the tidal classification was used for ranking the relative impact of the

tidal range on historic coastal landfill sites in this research. The severity score associated with the tidal classification was assigned as shown in Table 5.3. The severity scores are a simplified version of those used in the multi-scale coastal vulnerability index for Northern Ireland (McLaughlin and Cooper, 2010) to allow the tidal classification to be used rather than more detailed tidal range data.

**Table 5.3: Determining the tidal classification severity score**

| Tidal range | Tidal environment | Tidal classification severity score |
|---|---|---|
| <2 m | Microtidal | 5 |
| 2-4 m | Mesotidal | 3 |
| >4 m | Macrotidal | 1 |

Severity scores adapted from McLaughlin and Cooper (2010)

#### 5.3.3.3 Flooding

Flooding increases the probability of landfills eroding both due to the movement of water over the site (Laner et al., 2008b) and because infiltration of high volumes of water can adversely affect the structural integrity of the waste (Blight and Fourie, 2005). In addition, the build-up of water pressure behind a flood defence can cause it to fail exposing waste (Cooper et al., 2013). The Environment Agency publishes two versions of the fluvial and coastal flood maps. The first, the Flood Map for Planning (Rivers and Sea) (Environment Agency, 2016c) shows Flood Zones 2 and 3, and (most) flood defences, but does not take flood defences into account when determining flood zones. This was used to define historic coastal landfills, (i.e. those in tidal Flood Zone 3, see Chapter 2). The second map, Risk of Flooding from Rivers and Sea (RoFRS) (Environment Agency, 2016e), shows the residual flood zones after mitigation by flood defences broken into four categories: Very Low (annual probability <0.1%), Low (0.1% ≤ annual probability <1%), Medium (1% ≤ annual probability < 3.3%), and High (3.3% ≤ annual probability) (Environment Agency, 2013e). Both are available as GIS datasets for EA partners (UK Government, 2016).

Severity scores for flooding were determined by measuring the proportion of the landfill site flooded under each annual probability of flooding in RoFRS using GIS software (esri ArcMap), the annual probability of flooding that covered the greatest proportion of the site was used to assign the severity score as per Table 5.4.

**Table 5.4: Determining the flooding severity score**

| Predominant RoFRS category | Flooding severity score |
|---|---|
| None | 1 |
| Very low | 2 |
| Low | 3 |
| Medium | 4 |
| High | 5 |

Flood maps for reservoirs (Environment Agency, 2016d), pluvial (Environment Agency, 2016f) and groundwater flooding (UK Government, 2016) are also available, but as these types of flooding are relatively low energy events they were not included in the assessment.

### 5.3.4 Coastal landfill vulnerability related data available for England

The main factors to be considered when rating landfill vulnerability are the position, topography and condition of the landfill sites, the presence/absence of flood defences and the probability of defence failures, coastal slope, sediment balance and the presence of any buffer zones, e.g. saltmarshes (see sections 5.3.1 and 5.3.2). The best available data that either measure these directly or are suitable proxies are considered below. The severity scores assigned for the landfill position, exposed boundary length, defence type, sediment balance and buffer zone parameters are new for this research and are provisional pending future refinement of the risk screening assessment through consultation with relevant experts.

#### 5.3.4.1 Position of the landfill sites and exposed boundary length

The closer the landfill is to mean high water, the greater the risk of it being eroded. The Environment Agency's historic landfill GIS database is available to download under licence (Environment Agency, 2015b) and shows the landfill site boundaries. There are a number of GIS datasets that show the width of rivers along their tidal extent, e.g. (mean) High Water line in the OS Boundary-Line (Ordnance Survey, 2016) and WFD Transitional and Coastal Waterbodies Cycle 2 (Environment Agency, 2016a). There are discrepancies in the position of the high water line between different datasets due to different update frequencies and scales used. This research used the (mean) High Water line in the OS Boundary-Line dataset as it was the most recently updated of the large

scale datasets (1:10,000) and OS data are used to produce the WFD dataset (Environment Agency, 2016a). The distance between landfill boundaries and mean high water shown in the OS Boundary-Line GIS dataset (Ordnance Survey, 2016) was used to assign a landfill position severity score based in accordance with Table 5.5.

**Table 5.5: Determining the landfill position severity score**

| Distance from landfill boundary to mean high water | Landfill position severity score |
|---|---|
| distance ≤ 5 m | 5 |
| 5 m < distance ≤ 20 m | 4 |
| 20 m < distance ≤ 35 m | 3 |
| 35 m < distance ≤ 50 m | 2 |
| distance > 50 m | 1 |

The length of the landfill boundary exposed to wave impact, which can be determined from the GIS database of landfill sites (Environment Agency, 2013f; 2015b), will also influence the probability of waste being eroded, e.g. the two sites shown in Figure 5.4 are identical in size, but landfill site B is has a greater cross-sectional area exposed to wave impact and, therefore, is more likely to be subject to coastal erosion than landfill site A if other factors are identical. The length of the landfill boundary at the shoreline side of the landfill was used to assign each landfill's exposed boundary length severity score in accordance with Table 5.6.

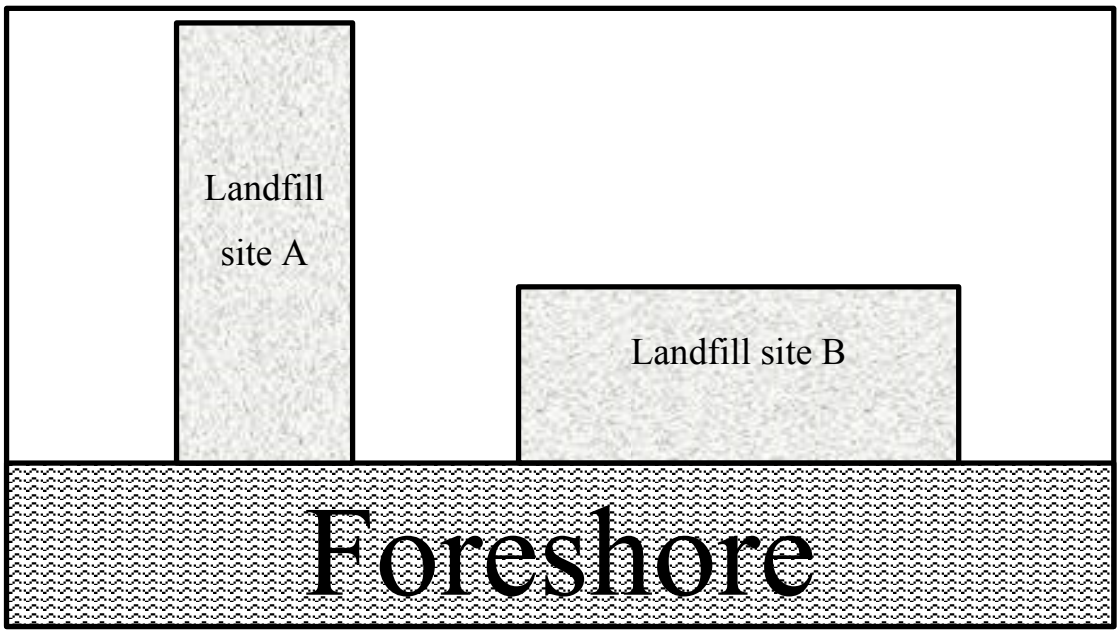



**Figure 5.4: The length of the landfill site boundary exposed to wave action will influence waste erosion rates**

**Table 5.6: Determining the exposed boundary length severity score**

| Length of landfill boundary facing the foreshore | Exposed boundary length severity score |
|---|---|
| length ≤ 500 m | 1 |
| 500 m < length ≤ 1000 m | 2 |
| 1000 m < length ≤ 2000 m | 3 |
| 2000 m < length ≤ 3000 m | 4 |
| length > 3000 m | 5 |

### 5.3.4.2 Condition of the historic coastal landfills

All of the sites considered by this research are closed landfills that have been capped, but some have already eroded and are releasing waste, e.g. Tilbury in Essex (Plate 5.1), or have waste migrating to the surface, e.g. Two Tree Island in Essex (Plate 5.2). The Environment Agency database of historic landfill sites does not record their condition, e.g. erosion of capping materials, which would need to be determined by other desk study methods, e.g. local records, archive news reports, or through site visits, which are beyond the scope of a desktop based screening assessment. Therefore, the current condition of the sites was not included in the screening assessment.

**Plate 5.1: Eroded early 20th century landfill waste covers the beach at East Tilbury in Essex (J. Brand, 9/5/2016, at TQ 67459 75631)**

**Plate 5.2: Bin bags full of landfill waste are migrating to the surface of Two Tree Island in Essex (J. Brand, 9/8/2015, at TQ 81439 85016)**

#### 5.3.4.3 Presence of flood defences and the probability of breaches

One-third of the English and Welsh coastline has shoreline defences (de la Vega-Leinert and Nicholls, 2008). The presence of flood defences with a design standard of at least 1 in 100 for fluvial flooding or 1 in 200 for tidal flooding can be determined using the EA's Spatial Flood Defences GIS dataset, which is available to EA partners (UK Government, 2016). Analysis of the probability of coastal defences breaching requires large amounts of data that are not available for all defences (Hall et al., 2006c). The EA uses a set of standardised fragility curves to assess the probability of flood defences breaching under different loadings; however, they only consider a limited number of the main defence failure modes (compare Table 5.7 and Table 5.8), and it is recommended to develop site specific fragility curves when prioritising site management (Simm et al., 2008). In addition, the existing fragility curves were not developed for use where landfill waste forms part of the defence structure (Environment Agency, 2008), and currently there is insufficient understanding of the mechanical properties of waste to enable additional fragility curves to be developed (Dixon and Jones, 2005; Gomes et al., 2014).

The probability of defences breaching is linked to the probability of them overtopping and coastal erosion (Table 5.7), which are already accounted for within this (the landfill vulnerability) sub-index. The probability of defences breaching is also linked to their current state of repair (Table 5.7) and to the defence type: soft engineered structures, e.g. earth embankments, are considered more likely to breach than hard engineered structures, e.g. seawalls, and to a greater width (Scott Wilson, 2008; Environment Agency, 2010c).

The state of repair of defences is based on a visual inspection of the site in accordance with the Environment Agency's Condition Assessment Manual (Environment Agency, 2006). Defences are assigned condition grades that range from 1=Very Good to 5=Very Poor (see Table 5.9), the higher the grade the greater the probability of the defence failing. Defence types and condition grades are recorded in the EA's Spatial Flood Defences GIS dataset (UK Government, 2016). This research used the defence type and condition as proxies for the probability of defence failure. The condition grade was used directly as a defence condition severity score, with the highest severity score of 5 also applied to sites where no flood defence exists. The defence type was scored as shown in

Table 5.10. Where the landfill site forms the flood defence the landfill site was not considered to be protected by a flood defence.

**Table 5.7: Key failure modes for flood defences**

| Defence type | | Key failure modes |
|---|---|---|
| Embankment/sloping seawall | | • Erosion of crest and inside face leading to breach following overtopping (possibly induced by settlement)<br>• Piping, excessive seepage, breach or collapse following deterioration due to vermin infestation<br>• Breach following failure of foreign objects or weak spots caused by their presence |
| Slope protection against erosion | | • Structural failure following vandalism<br>• Toe erosion/foundation failure<br>• Slip failure due to instability or foundation failure<br>• Failure of slope drainage<br>• Damage by boats and barges<br>• Structural failure of inflexibility of rigid revetments placed on dynamic watercourses/coastlines |
| Vertical wall structures | | • Overtopping<br>• Toe erosion<br>• Failure of structural members (e.g. tie-rod or anchorage system)<br>• Structural failure due to wash out of fill following joint failure<br>• Structural failure following abrasion or corrosion |
| Beaches | Sand/Shingle beach | Beach roll-back and erosion are natural cyclic processes rather than failure<br>Beaches fail when they do not perform their primary function (e.g. overtopping/ tidal flooding/erosion protection), although they may recover with time.<br>Key processes resulting in failure:<br>• Overtopping due to erosion/gullying/reduced energy dissipation following beach lowering<br>• Failure of control structures |

Source: Defra and Environment Agency (2007b, p. 3)

**Table 5.8: Failure modes for flood defences used for fragility curve development**

| Flood and coastal defence type | | Indicative failure modes |
|---|---|---|
| Embankment/sloping seawall | | • Erosion of crest and inside face leading to breach following overtopping<br>• Piping, excessive seepage (river embankments) |
| Vertical wall structures | Anchored sheet pile | • Toe erosion leading to rotation about the tie rod<br>• Rupture of the tie rod following toe erosion and corrosion |
| | Cantilever wall | • Scour on the toe of the sheet pile followed by instability and collapse of the wall |
| | Masonry wall | • Overturning of the structure<br>• Sliding of the structure |
| Sand/Shingle beach | | • Breaching of the beach after crest retreat |

Source: Defra and Environment Agency (2007b, p. 4)

**Table 5.9: Condition grades used by the Environment Agency when assessing flood defence assets in accordance with the Condition Assessment Manual**

| Grade | Rating | Description |
|---|---|---|
| 1 | Very good | Cosmetic defects that will have no effect on performance |
| 2 | Good | Minor defects that will not reduce the overall performance of the asset |
| 3 | Fair | Defects that could reduce performance of the asset |
| 4 | Poor | Defects that would significantly reduce the performance of the asset. Further investigation needed |
| 5 | Very poor | Severe defects resulting in complete performance failure |

Source: Environment Agency (2006, p. 10)

**Table 5.10: Determining the defence type severity score**

| Defence type | Defence type severity score |
|---|---|
| Hard defences | 1 |
| Mixture of hard and soft defences | 2 |
| Soft defences | 3 |
| Some defences, but not all of the water facing boundary protected | 4 |
| No defences present or the landfill is the defence | 5 |

#### 5.3.4.4 Coastal slope

Researchers consider coastal slope differently depending on the focus of the coastal vulnerability assessment, for example Palmer *et al*. (2011) are concerned with the risk

of coastal erosion and consider beach slope, using beach width as a proxy, and the distance from the back of the beach to the 20 m isobaths, whereas Khouakhi *et al*. (2013) are concerned with the risk of damage to property and only consider the slope above the shoreline (using the distance between the coastline and the 10 m inland elevation). No free GIS dataset could be found to determine isobaths at an appropriate scale for assessing slope for individual landfill sites. The British Geological Survey DigBath250 GIS dataset (British Geological Survey, 2016a) includes the 20 m isobaths and would be the most appropriate dataset for assessing coastal slope, but was beyond the budget of this research. Therefore, the Portal for Bathymetry online map (European Marine Observation and Data Network, 2016) depth profile function was used to approximate distances between landfills and the 20 m isobaths. These distances were then used to assign the coastal slope severity score in accordance with Table 5.11. These severity scores are based on Palmer *et al*. (2011), but an additional severity score has been added to refine the scoring as their scoring method only used four categories (<1 km, 1 to 2 km, 2 to 4 km, and > 4km).

**Table 5.11: Determining the coastal slope severity score**

| Distance from landfill boundary to 20 m isobath | Coastal slope severity score |
|---|---|
| distance ≤ 1 km | 5 |
| 1 km < distance ≤ 2 km | 4 |
| 2 km < distance ≤ 3 km | 3 |
| 3 km < distance ≤ 4 km | 2 |
| distance > 4 km | 1 |

After Palmer et al. (2011)

#### 5.3.4.5 Sediment balance

Some Shoreline Management Plans include coastal erosion maps that show whether areas are eroding or accruing material (e.g. Royal Haskoning, 2009; Environment Agency, 2010a), others include similar information in the text or tables, which are impracticable to cross-reference to landfill locations for a large-scale assessment. The SMPs also include maps showing predictions of coastal erosion for three defined time periods (20, 50 and 100 years with a baseline of 2005) both if there is No Active Intervention (NAI) and if the current defence maintenance regime is continued for each Policy Unit. These predictions take into account climate change effects, coastal geology

and the current condition of the defences. The With Present Management (WPM) erosion predictions also include consideration of the residual life expectancy of the flood defences under the current maintenance regime. The small scale of the SMP maps makes it impossible to determine erosion rates at landfill locations accurately and GIS datasets for them could not be obtained, but they are suitable for providing an indication of whether sediment is accreting or eroding at landfill locations. This research used the WPM erosion predictions as a proxy to assign the sediment balance severity score in accordance with Table 5.12. The WPM maps were chosen rather than the NAI maps as the WPM maps account for any artificial sediment recharge that may be taking place.

**Table 5.12: Determining the sediment balance severity score**

| Sediment balance | Sediment balance severity score |
|---|---|
| Accretion | 1 |
| No change | 3 |
| Erosion | 5 |

#### 5.3.4.6 Buffer zones

The presence of vegetated saltmarshes can significantly attenuate the impact of waves upon flood defences, dissipating up to half of the wave energy in the first 10-20 metres of saltmarsh surface, reducing the risk of defences being overtopped or breached (Möller and Spencer, 2002; Committee on Climate Change, 2013). A GIS dataset of saltmarsh extents is available to EA partners (UK Government, 2016) and was used to determine the average width of saltmarsh in front of the landfill. The average saltmarsh width was used to assign a severity score as per Table 5.13.

**Table 5.13: Determining the buffer zone severity score**

| Width of saltmarsh | Buffer zone severity score |
|---|---|
| No saltmarsh | 5 |
| 0 m < width ≤ 10 m | 4 |
| 10 m < width ≤ 20 m | 3 |
| 20 m < width ≤ 50 m | 2 |
| width > 50 m | 1 |

### 5.3.5 Calculation of the coastal drivers and landfill vulnerability sub-indices

The selected parameters for the coastal drivers and the landfill vulnerability sub-indices are summarised in Table 5.14 and Table 5.15 respectively. When summed, the parameters selected result in a coastal drivers sub-index value ranging from 3 to 15 and a landfill vulnerability sub-index value ranging from 7 to 35. Therefore, the formulae to normalise these values to percentages to enable the sub-indices to be combined into the overall risk index (after McLaughlin and Cooper, 2010) are as shown in Equation 5.6 and Equation 5.7.

**Table 5.14: Summary of coastal drivers sub-index parameters severity scoring**

| **Parameters** | **Severity scores** |
|---|---|
| Wave energy | 1: fetch < 10 km or saltmarsh present<br>3: 10 km ≤ fetch ≤ 100 km<br>5: fetch > 100 km |
| Tidal classification | 1: macrotidal<br>3: mesotidal<br>5: microtidal |
| Flooding | 1: landfill is predominantly outside of RoFRS flood risk zones<br>2: predominant RoFRS category is very low<br>3: predominant RoFRS category is low<br>4: predominant RoFRS category is medium<br>5: predominant RoFRS category is high |
| **Minimum coastal drivers sub-index value = 3**<br>**Maximum coastal drivers sub-index value = 15** | |

**Table 5.15: Summary of landfill vulnerability sub-index parameters severity scoring**

| Parameters | Severity scores |
|---|---|
| Landfill position | Landfill boundary to mean high water:<br>1: distance > 50 m<br>2: 35 m < distance ≤ 50 m<br>3: 20 m < distance ≤ 35 m<br>4: 5 m < distance ≤ 20 m<br>5: distance ≤ 5 m |
| Exposed boundary length | Length of landfill boundary facing foreshore:<br>1: length ≤ 500 m<br>2: 500 m < length ≤ 1000 m<br>3: 1000 m < length ≤ 2000 m<br>4: 2000 m < length ≤ 3000 m<br>5: length > 3000 m |
| Defence condition | 1: condition grade is 1<br>2: condition grade is 2<br>3: condition grade is 3<br>4: condition grade is 4<br>5: condition grade is 5 or no defence present |
| Defence type | 1: hard defences<br>2: mixture of hard and soft defences<br>3: soft defences<br>4: some defences, but not all of the boundary protected<br>5: no defences present or the landfill is the defence |
| Coastal slope | Distance between landfill and 20 m isobath:<br>1: distance > 4 km<br>2: 3 km < distance ≤ 4 km<br>3: 2 km < distance ≤ 3 km<br>4: 1 km < distance ≤ 2 km<br>5: distance ≤ 1 km |
| Sediment balance | 1: accretion<br>3: no change<br>5: erosion |
| Buffer zone | Width of saltmarsh:<br>1: width > 50 m<br>2: 20 m < width ≤ 50 m<br>3: 10 m < width ≤ 20 m<br>4: 0 m < width ≤ 10 m<br>5: No saltmarsh |
| **Minimum landfill vulnerability sub-index value = 7**<br>**Maximum landfill vulnerability sub-index value = 35** | |

**Equation 5.6: Calculation of the normalised coastal drivers sub-index**

$$Normalised\ coastal\ driver\ sub\text{-}index = \frac{\sum severity\ scores - 3}{12}\ x\ 100$$

**Equation 5.7: Calculation of the normalised landfill vulnerability sub-index**

$$Normalised\ landfill\ vulnerability\ sub\text{-}index = \frac{\sum severity\ scores - 7}{28}\ x\ 100$$

## 5.4 Selection of parameters to determine the landfill hazard sub-index

### 5.4.1 Hazard related parameters used in landfill risk assessments

Existing landfill risk ranking methods use a maximum of four hazard related parameters, see Table 5.16. Data available for England for the parameters are indicated in the table and explored in more detail in section 5.4.2. Parameters used depend on the overall aim of the specific method and can be summarised as quantities and types of waste parameters, and contaminant concentration parameters. Only the Austrian (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013) and WEAR (Alaska Department of Environmental Conservation, 2015) methods consider the hazard from eroded waste, the other methods only consider the pollution potential of fully enclosed waste, e.g. contamination of groundwater by leachates. Both the Austrian and WEAR methods assume the total mass of the landfilled waste will be eroded when determining the hazard and do not consider the rate at which material is released. The Austrian method only considers the hazard from soluble metals that have entered the water column and calculates the expected metal release based on generic initial concentrations of metals in the waste at the time of landfilling, the age of the waste and generic leachable fractions. The WEAR method does not consider contaminant concentrations in the waste, instead it ranks the hazard based on the site type.

**Table 5.16: Source parameters used in existing landfill risk ranking methods**

| Source parameters | Method | | | | | | Existing datasets available for sites in England? |
|---|---|---|---|---|---|---|---|
| | LPI | ELI | GW | Serbian | Austrian | WEAR | |
| Site status, e.g. managed/unmanaged, open/closed, waste covered/not covered | | ✓ | | | | ✓ | Yes-except exposure of waste due to erosion |
| Site size, e.g. area or volume | | | ✓ | ✓ | ✓ | ✓ | Area, some volume |
| Operating period, e.g. 1960 to 1965 | | | | | ✓ | ✓ | 49% of sites |
| Contaminant concentrations in waste | | | | | ✓ | | No |
| Site/waste type, e.g. MSW, industrial | | | ✓ | | | ✓ | 82% of sites |
| Waste composition, i.e. material types | | ✓ | | | | | No |
| Contaminants in leachates | ✓ | ✓ | | | ✓ | | Some sites |
| Precipitation and infiltration → leachate volume | | | ✓ | | | | Precipitation only |

LPI = Leachate pollution index (Kumar and Alappat, 2005)

ELI = Environment-landfill interaction index (Calvo et al., 2007)

GW = Groundwater contamination hazard rating for landfills (Singh et al., 2009)

Serbian = Prioritisation of landfills subject to flood risk in Serbia (Okaneya et al., 2013)

Austrian = Austrian fluvial flooding of landfills risk assessment (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013)

WEAR = Waste Erosion Assessment and Review project (Alaska Department of Environmental Conservation, 2015)

## 5.4.2 Hazard related data available for England

The best available data that either measure the quantities of waste, types of waste and contaminant concentrations directly or are suitable proxies are considered below. The severity scores assigned for the landfill volume, landfill type, salinity and dissolved contaminant parameters are new for this research and are provisional pending future refinement of the risk screening assessment through consultation with relevant experts.

### 5.4.2.1 Total volumes of waste and rate of erosion

Existing landfill risk ranking methods (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013; Alaska Department of Environmental Conservation, 2015) determine the hazard posed by eroded waste by assuming the entire landfill will erode, this assumption is presumably made as saturated waste is known to be mechanically unstable (Blight and Fourie, 2005; Liang et al., 2015). The area of historic landfill sites can be determined using the national database (Environment Agency,

2015b) and GIS mapping software, e.g. ArcMap; however, the database does not provide information on waste volumes. A survey of coastal county councils around England conducted for this research found that Devon, Essex, Northumberland, North Yorkshire and Suffolk County Councils have records that estimate volumes of waste for at least some of the historic coastal landfills in their counties, but Kent, Hampshire and Somerset County Councils do not (Table 5.17). Other county councils only provided site location data or did not respond. Where records do not include the landfilled volume it may be possible to estimate the depth of the waste using site design drawings (where available) or records of any previous invasive site investigations that determined the waste depth. Alternatively, the landfill's depth can be approximated by comparing historic records on site topography before landfilling compared to its present topography, e.g. historical Ordnance Survey maps available on the internet (National Library of Scotland, 2016) or from monitoring well depths (where installed). In some landfill locations BGS borehole records (British Geological Survey, 2016b) may exist that either pre-date the landfill and, hence show the natural ground level, or were drilled through the landfill and record the waste depth. The landfill depth combined with the site boundary information from the EA historic landfill database can then be used to approximate the landfilled volume.

A potentially more accurate method of determining the waste's depth would be to use geophysical surveying techniques; continuous surface wave seismic surveys are the most commonly used geophysics method to determine landfill depths (Hutchinson and Barta, 2000; Styles and Cassidy, 2009; Reynolds, 2011). However, geophysical surveys require significant resources to undertake and are beyond the scope of a desk-based screening study.

The total volume of the landfill, which represents the maximum amount of material that could be released, has been assigned a landfill volume severity score as per Table 5.18. The 2,000,000 $m^3$ threshold for the highest severity score category was set with the aim that circa 20% of historic coastal landfill sites would exceed it, based on estimated volumes for all of the historic coastal landfill sites calculated using their surfaces areas from GIS datasets multiplied by an assumed average depth of 15 m. The assumed average depth of 15 m was selected as only the very largest landfills in the UK are over 30 m deep (Robinson, 2007).

**Table 5.17: Data held by coastal county councils regarding historic coastal landfill sites around the coast of England. Other coastal county councils only provided location data or did not respond**

| | County Council holds records relating to the parameter? | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Landfill parameter** | **Devon** | **Essex** | **Hampshire** | **Kent** | **Northumberland** | **North Yorkshire** | **Somerset** | **Suffolk** |
| Operating dates | Yes | Yes | No | Yes | Yes | Yes | No | Yes |
| Site type | Yes | Yes | No | Yes | Yes | Yes | No | Yes |
| Site area | Yes | Yes | No | Yes | No | Yes | No | No |
| Average waste depth | No | Yes | No | No | No | Yes | No | Yes |
| Waste mass or volume | Yes | Yes | No | No | Yes | Yes | No | Yes |
| Environmental controls, e.g. leachate management | Yes | Yes | No | Yes | Yes | Yes | No | Yes |
| Environmental monitoring | Yes | Yes | No | Yes | Yes | Yes | Yes | No response |
| References (all pers. comm. by email) | S. Price, 18/12/2013 | A. Brown, 26/10/2015 | D. Emmett, 6/1/2014. A. Galea, 8/1/12014 | L. Pronger, 23/12/2013 | S. Wardle, 17/12/2013 | A. Surman, 20/1/2014 | D. Oaten, 6/1/2014 | Anon. Suffolk County Council, 8/1/2014 |

**Table 5.18: Determining the landfill volume severity score**

| Landfill volume ($m^3$) | Landfill volume severity score |
|---|---|
| volume ≤ 500,000 | 1 |
| 500,000 < volume ≤ 1,000,000 | 2 |
| 1,000,000 < volume ≤ 1,500,000 | 3 |
| 1,500,000 < volume ≤ 2,000,000 | 4 |
| volume > 2,000,000 | 5 |

Although existing landfill risk methods assume the entire waste volume will erode, it seems unlikely that this would be the case for larger landfills around the coast of the UK as waste is often deposited in discrete cells and the materials dividing the cells are likely to be more resilient to erosion than the waste. In addition, if the waste release is related to a flood defence failure it is likely that the flood defence will be repaired before the

entire waste volume has been eroded. Therefore, to assess the magnitude of the hazard from eroded waste materials, consideration also needs to be given to how quickly waste materials are likely to erode from sites once their capping materials have been breached as well as how much waste is present overall. The current limited understanding of the mechanical properties of landfilled waste restricts the use of erosion models for these sites (Dixon and Jones, 2005; Gomes et al., 2014) so it is necessary to use proxies that represent the likely rate of release. With the exception of the mechanical stability of the waste materials for which there are no data, the factors that will determine the rate that material would erode will be those that determine the wave energy reaching the exposed waste. Therefore, the following six parameters were also used in determining the landfill hazard sub-index using the same severity scores as determined for the waste release index: wave energy, tidal classification, landfill position, defence type, coastal slope and (vegetated) buffer zones (see sections 5.3.3 and 5.3.4).

#### 5.4.2.2 Contaminant concentrations in eroded waste

The historic landfill database provides an indication of whether sites are believed to contain inert, industrial, commercial, household, special waste, liquids/sludge or if the type of waste is unknown. Just 37% of the sites are believed to contain only a single waste type, 45% of the sites contain a mixture of waste types in unknown proportions and 18% of the sites have no record of the waste type. The mixture of materials in the waste is related to the waste types that were landfilled, but even where the waste type is recorded, it is known that the mixture of materials and contaminant concentrations within waste types vary depending on when the waste was deposited (Parfitt, 2009; Quaghebeur et al., 2013). The database contains the opening date of 823 of the 1264 historic coastal landfills and the closing date of 623 of them; therefore, the operating periods of 51% of the 1264 historic coastal landfill sites are not recorded in the national database. Waste types and operating periods may be available in the archives of local authorities or individual site operators, or may not have been kept due to less stringent regulations on record keeping at the time the sites were operating. A survey of coastal county councils around England conducted for this research found that Devon, Essex, Kent, Northumberland, North Yorkshire and Suffolk County Councils have records of operating dates and site types for at least some of the historic coastal landfills in their counties, but Hampshire and Somerset County Councils do not (Table 5.17).

Even where the waste type and operating period is known it only provides an indication of the types of materials that may be present within a site, it does not provide information on contaminant concentrations, which are required in order to accurately determine the potential for pollution from eroding materials. As seen in Chapter 3, contaminant concentrations in landfill sites are highly variable (differing by up to four orders of magnitude between minimum and maximum concentrations at each sampling location), it is not possible to predict contaminant concentrations in the waste without sampling and analysing it, and even after analysis it is still not known whether the measured contaminant concentrations are representative of the whole site. Contaminant concentrations on different materials within individual sites vary significantly and materials in landfill sites are not well mixed, therefore contaminant release could vary significantly depending upon which materials erode. In addition, as seen in Chapter 3, contaminant concentrations can vary by at least an order of magnitude between sites. Even if the contaminant concentrations from one site could be used to predict concentrations in another site, the availability of the contaminants will depend upon speciation and the types of particle they are sorbed to/associated with, which would need to be assessed at a micro-scale for individual landfill sites (Neuhold, 2013) and is impracticable for a national screening assessment of multiple sites. Therefore, to use standardised contaminant concentrations for all landfill sites under consideration as used by some existing methods (e.g. Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013) is inaccurate and potentially misleading to the end-user.

The maximum permissible (leachable) concentrations of contaminants in materials being landfilled vary with the landfill site type, e.g. sites that are permitted to take hazardous waste (also known as special waste) are allowed maximum (leachable) concentrations of mercury 200 times higher, and of chromium 140 times higher, than inert sites (Council Decision, 2003). Therefore, for this research the WEAR method's approach was used where the site type is used for ranking the severity of the hazard from contaminants in the waste (Alaska Department of Environmental Conservation, 2015), which for the purposes of a screening assessment is considered an appropriate proxy (Singh et al., 2009). The severity of the hazard is known to increase in the site type order: inert, e.g. construction and demolition waste; MSW or commercial; industrial; special waste (NetRegs, nd; Council Decision, 2003; Singh et al., 2009;

Alaska Department of Environmental Conservation, 2015). No information could be found relating to how hazardous liquids/sludge are in relation to other waste types and the unknown waste could contain any combination of the known waste types; therefore, liquids/sludge and unknown landfill types have both been assigned the highest hazard rating. Severity scores were assigned for the different site types as per Table 5.19. Where the site has more than one type assigned to it in the landfill database the type with the highest severity score was used to determine the landfill's overall severity score.

**Table 5.19: Determining the landfill type severity score**

| Landfill type | Landfill type severity score |
|---|---|
| Inert | 1 |
| MSW or commercial | 2 |
| Industrial | 3 |
| Special waste | 4 |
| Liquids/sludge or unknown | 5 |

#### 5.4.2.3 Release of contaminants from eroded waste to the water column

The proportion of metals released from waste eroded into water varies significantly between sites (see Chapter 4) and, therefore, contrary to the Austrian risk assessment method (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013), metal release cannot be predicted using a standard proportion even if concentrations of metals within the waste are known. Leachates from some landfills are monitored (Table 5.17) but not all, and the proportions of metals released from eroded waste are likely to be higher than fully contained waste as there will be no attenuation within surrounding capping materials and sediments (Kjeldsen et al., 2002). In addition, leaching of metals from waste in saline environments is significantly higher than in freshwater environments (see Chapter 4), so any existing leachate data are unlikely to be representative of metal release due to waste erosion into tidal waters. Conversely, except in some rare cases, organic compounds are less soluble in saline waters than freshwaters due to the 'salting-out' effect (Xie et al., 1997; Kim and Osako, 2003), so any existing leachate data are unlikely to be representative of persistent organic pollutant release due to waste erosion in tidal waters.

Therefore, there are insufficient data to determine the release of contaminants into the water column. However, it is possible to rank the hazard posed by leached contaminants using the amount of waste eroded (see 5.4.2.1) and contaminant concentrations in the waste (see 5.4.2.2) as proxies for the maximum mass of contaminants that could leach, and by considering the salinity of, and dilution in, the receiving waters.

Inorganic contaminant release is significantly higher in saline waters compared to freshwaters (see Chapter 4); consequently, concentrations of contaminants in the receiving waters will be affected by the salinity of the water the waste has eroded into. Therefore, it is necessary to include salinity as a parameter in the risk screening assessment. The Joint Nature Conservation Council's (JNCC) website includes a downloadable Variable Salinity Areas dataset that shows oligohaline (0.5-5 ppt salinity), mesohaline (5-18 ppt salinity) and polyhaline (18-30 ppt salinity) zones around the coast of the UK (McBreen et al., 2011). These zones in combination with the landfill location maps were used to assign salinity severity scores as per Table 5.20. Where a landfill was bordered by more than one salinity zone, the highest severity score of the bordering zones was used.

**Table 5.20: Determining the salinity severity score**

| Salinity zone | Salinity severity score |
|---|---|
| upstream of oligohaline zone | 1 |
| oligohaline zone | 2 |
| mesohaline zone | 3 |
| polyhaline zone | 4 |
| downstream of polyhaline zone | 5 |

Dilution of contaminants in the water column will depend on the volume of water acting as a leachant, which would be the effective volume of the waterbody the waste has eroded into. In estuaries the effective volume is made up of contributions from river flow and tidal exchange. Tidal prism volumes can be calculated at an accuracy appropriate to rank the available dilution in different estuaries from tidal exchange using the tidal range and estuary's surface area (Equation 5.8). The tidal range is given in the SMPs and the estuarine surface area can be determined using GIS software to measure the transitional area in the WFD transitional and coastal waterbodies cycle 2 dataset

(UK Government, 2016). For landfill sites in estuaries, the tidal prism was used in this research as a proxy for the total effective volume of water acting as a leachant and (dissolved contaminant) severity scores were assigned as per Table 5.21. Sites upstream of the tidal limit were given the highest severity score as they would have the least dilution and sites on coastal waters were given the lowest severity score on the assumption that coastal waters would have the greatest dilution. For estuaries large enough to be split into multiple transitional zones in the WFD dataset, only the zone adjacent to the landfill was assumed to contribute to dilution of any released contaminants when calculating the tidal prism volume.

**Equation 5.8: Calculation of tidal prism volume (after Mikhailov and Mikhailova, 2012)**

$$V = T \, x \, A$$

*Where:*
*$V$ = calculated tidal prism volume ($m^3$)*
*$T$ = annual extreme tidal range averaged over the estuary ($m$)*
*$A$ = measured transitional area of the estuary ($m^2$)*

**Table 5.21: Determining the dissolved contaminant severity score**

| Tidal prism ($m^3$) | Dissolved contaminant severity score |
|---|---|
| Upstream of tidal limit or tidal prism ≤ 5,000,000 | 5 |
| 5,000,000 < Tidal prism ≤ 50,000,000 | 4 |
| 50,000,000 < Tidal prism ≤ 100,000,000 | 3 |
| 100,000,000 < Tidal prism ≤ 500,000,000 | 2 |
| Tidal prism > 500,000,000 or open coast | 1 |

### 5.4.3 Calculation of the landfill hazard sub-index

The selected parameters for the landfill hazard sub-index are summarised in Table 5.22. When summed, the parameters selected result in a landfill hazard sub-index value ranging from 10 to 50. Therefore, the formula to normalise this score to a percentage to enable the sub-indices to be combined into the overall risk index (after McLaughlin and Cooper, 2010) is as shown in Equation 5.9.

**Table 5.22: Summary of landfill hazard sub-index parameters severity scoring**

| Parameters | Severity scores |
|---|---|
| Landfill volume | 1: volume ≤ 500,000 $m^3$<br>2: 500,000 $m^3$ < volume ≤ 1,000,000 $m^3$<br>3: 1,000,000 $m^3$ < volume ≤ 1,500,000 $m^3$<br>4: 1,500,000 $m^3$ < volume ≤ 2,000,000 $m^3$<br>5: volume > 2,000,000 $m^3$ |
| Wave energy | 1: fetch < 10 km or saltmarsh present<br>3: 10 km ≤ fetch ≤ 100 km<br>5: fetch > 100 km |
| Tidal classification | 1: macrotidal<br>3: mesotidal<br>5: microtidal |
| Landfill position | Landfill boundary to mean high water: 1: distance > 50 m<br>2: 35 m < distance ≤ 50 m<br>3: 20 m < distance ≤ 35 m<br>4: 5 m < distance ≤ 20 m<br>5: distance ≤ 5 m |
| Defence type | 1: hard defences<br>2: mixture of hard and soft defences<br>3: soft defences<br>4: some defences, but not all of the boundary protected<br>5: no defences present or the landfill is the defence |
| Coastal slope | Distance between landfill and 20 m isobath: 1: distance > 4 km<br>2: 3 km < distance ≤ 4 km<br>3: 2 km < distance ≤ 3 km<br>4: 1 km < distance ≤ 2 km<br>5: distance ≤ 1 km |
| Buffer zone | Width of saltmarsh: 1: width > 50 m<br>2: 20 m < width ≤ 50 m<br>3: 10 m < width ≤ 20 m<br>4: 0 m < width ≤ 10 m<br>5: No saltmarsh |
| Landfill type | 1: Inert<br>2: MSW or commercial<br>3: Industrial<br>4: Special waste<br>5: Liquids/sludge or unknown |
| Salinity | 1: upstream of oligohaline zone<br>2: oligohaline zone<br>3: mesohaline zone<br>4: polyhaline zone<br>5: downstream of polyhaline zone |
| Dissolved contaminant | 1: Tidal prism > 500,000,000 $m^3$ or open coast<br>2: 100,000,000 $m^3$ < Tidal prism ≤ 500,000,000 $m^3$<br>3: 50,000,000 $m^3$ < Tidal prism ≤ 100,000,000 $m^3$<br>4: 5,000,000 $m^3$ < Tidal prism ≤ 50,000,000 $m^3$<br>5: Upstream of tidal limit or tidal prism ≤ 5,000,000 $m^3$ |
| **Minimum landfill hazard sub-index value = 10**<br>**Maximum landfill hazard sub-index value = 50** | |

**Equation 5.9: Calculation of the normalised landfill hazard sub-index**

$$Normalised\ landfill\ hazard\ sub\text{-}index = \frac{\sum severity\ scores - 10}{40}\ x\ 100$$

## 5.5 Selection of parameters to determine the environmental vulnerability sub-index

### 5.5.1 Environmental vulnerability parameters used in existing landfill screening methods

Existing landfill risk ranking/prioritisation methods use a maximum of five receptor related parameters, see Table 5.23. Data available for England are indicated in the table. Parameters used depend on the overall aim of the specific method and include water use, the proximity of habitats, and the presence of flora and fauna (including humans). CIRIA C718 (Cooper et al., 2013) recommends the following receptors should be the main focus of risk assessments of historic coastal landfill sites in the UK:

- Humans (public health), i.e. site users, and users of neighbouring land.
- Designated environmental sites
    - Ecological systems: especially those legally protected, e.g. Ramsar, SAC, SPA (and candidate sites), SSSI, National Nature Reserves, Marine Nature Reserves (now Marine Conservation Zones), and Local Nature Reserves.
    - Other: National Parks and Heritage Coasts.
- Controlled waters, i.e. groundwater, lakes, rivers, canals and coastal waters.
- Property
    - Particularly relating to ignition of released gas, e.g. damage to buildings, infrastructure or Scheduled Monuments.
    - Crops or animals, domesticated or wild, that may be consumed by humans.

Data availability for parameters relevant for assessing the vulnerability of receptors in the intertidal zone and/or tidal waters around England are explored in section 5.5.2.

**Table 5.23: Receptor parameters used in existing landfill risk ranking methods**

| Receptor parameters | Method | | | | | | Existing datasets available for sites in England? |
|---|---|---|---|---|---|---|---|
| | LPI | ELI | GW | Serbian | Austrian | WEAR | |
| Drinking water source protection zone or water extraction points | | | ✓ | | | ✓ | Yes |
| Distance to critical habitat, e.g. protected area or area containing protected species | | | | | ✓ | ✓ | Yes |
| Distance to residential property | | | | | | ✓ | Yes |
| Distance to stressed habitat (field observation) | | | | | | ✓ | No |
| Distance to water course | | | | ✓ | | ✓ | Yes |
| Flora and fauna | | ✓ | | | | | Limited data |
| Groundwater use | | | ✓ | | | | Yes |

LPI = Leachate pollution index (Kumar and Alappat, 2005)

ELI = Environment-landfill interaction index (Calvo et al., 2007)

GW = Groundwater contamination hazard rating for landfills (Singh et al., 2009)

Serbian = Prioritisation of landfills subject to flood risk in Serbia (Okaneya et al., 2013)

Austrian = Austrian fluvial flooding of landfills risk assessment (Laner et al., 2008b; 2009; Neuhold and Nachtnebel, 2011; Neuhold, 2013)

WEAR = Waste Erosion Assessment and Review project (Alaska Department of Environmental Conservation, 2015)

## 5.5.2 Environmental vulnerability related data for England

The best available data that either measure the vulnerability of environmental receptors directly or are suitable proxies are considered below. The severity scores assigned for the human impact, designated sites and seafood parameters are new for this research and are provisional pending future refinement of the risk screening assessment through consultation with relevant experts.

### 5.5.2.1 Humans (public health)

In the intertidal zone, humans are most likely to come into contact with any eroded waste or released contaminants during recreational use of beaches. No dataset could be found that shows recreational beach use; however, an "areas affecting bathing waters" dataset showing catchments for bathing beaches is available to EA partners (UK Government, 2016). If concentrations of contaminants in the waste and how much waste is dispersed to beaches were known, Soil Guideline Values (SGVs) (Environment

Agency, 2009c) could potentially be used to determine the vulnerability of humans to eroded waste. However, SGVs cannot be used as there are insufficient data to determine contaminant concentrations (see 5.4.2.2) and dispersion modelling is beyond the scope of a screening assessment. In addition, the SGVs would not consider the potential for physical harm from contact with the waste or the potential for contaminants such as asbestos being present. Therefore, distances between bathing water catchments and historic coastal landfill sites were determined using GIS software and used as a proxy for the quantities of solid waste materials and dissolved contaminants that humans may come into contact with. This was based on the assumption that the greater the distance from the source of the waste, the greater the dispersion of the waste and dilution of the contaminants. The human impact severity scores used are shown in Table 5.24.

**Table 5.24: Determining the human impact severity score**

| Distance between bathing water catchment and landfill site | Human impact severity score |
|---|---|
| Landfill site falls within one or more bathing water catchment | 5 |
| 0 m < distance ≤ 50 m | 4 |
| 50 m < distance ≤ 100 m | 3 |
| 100 m < distance ≤ 150 m | 2 |
| distance > 150 m | 1 |

#### 5.5.2.2 Designated sites

There exists a multitude of environmentally designated sites around England. The availability of GIS datasets for those highlighted in CIRIA C718 and others that fall within the coastal (flood) zone are shown in Table 5.25. Also included in the table are heritage coasts, which are not designated sites, but are recommended for inclusion as a receptor by CIRIA C718 and were treated as designated sites for the purposes of this assessment. Distances between these sites and historic coastal landfills can easily be assessed on a national scale using GIS mapping software if distance ranges (buffer zones) are used rather than absolute distances to each site. These distances were used to determine (designated sites) severity scores for designated sites, which were assigned as per Table 5.26. Designated sites upstream of the landfills are included to account for tidal movement of contaminants. 1 km was selected as the upper buffer limit as it was agreed with regulators for use in a similar risk screening assessment as the distance within which drainage outfalls are considered to be a risk to protected sites, beyond

1 km it was considered that soluble pollutants will have been diluted to safe limits and sediment will have settled or dispersed (UK Government, 2009). However, further research is required to determine whether the same buffer limit is appropriate for contaminants originating from historic coastal landfills.

**Table 5.25: Available datasets for designated environmental sites and other ecological sites**

| Designated environmental sites | GIS dataset supplier | Publicly available? |
|---|---|---|
| **Ecology related designated sites** | | |
| Environmentally Sensitive Areas (ESAs) | Natural England (2016) | yes |
| Important Bird Areas (IBAs) | RSPB (2016) | yes |
| Local Nature Reserves (LNRs) | Natural England (2016) | yes |
| Marine Protected Areas (MPAs) includes SACs with Marine Components, SPAs with Marine Components, Marine Conservation Zones, Nature Conservation Marine Protected Areas (Scotland only), Marine Nature Reserves (Isle of Man only) and OSPAR MPAs | JNCC (2016) | yes |
| National Nature Reserves (NNRs) | Natural England (2016) | yes |
| OSPAR Marine Protected Areas | JNCC (2016) | yes |
| Priority Habitat Inventory | Natural England (2016) | yes |
| Recommended Marine Conservation Zones (rMCZs) | Natural England (2016) | yes |
| RAMSAR | JNCC (2016) | yes |
| RSPB Reserves | RSPB (2016) | yes |
| SAC (including SACs with Marine Components) | JNCC (2016) | yes |
| SPA (including SPAs with Marine Components) | JNCC (2016) | yes |
| SSSI | Natural England (2016) | yes |
| **Other designated environmental sites** | | |
| Areas of Outstanding Natural Beauty (AONB) | Natural England (2016) | yes |
| Country Park | Natural England (2016) | yes |
| National Parks | Natural England (2016) | yes |
| **Other environmental sites** | | |
| Heritage Coasts | Natural England (2016) | yes |
| NB All Natural England datasets are currently being moved as part of Defra's Open Data Programme and in the future will be downloadable from environment.data.gov.uk (UK Government, 2016) | | |

**Table 5.26: Determining the designated sites severity score**

| Distance between environmental/ecological site and landfill | Designated sites severity score |
|---|---|
| Landfill site is within one or more designated site(s) | 5 |
| Designated site within 100 m upstream or downstream of landfill | 4 |
| Designated site within 250 m upstream or 500 m downstream of landfill | 3 |
| Designated site within 250 m upstream or 1 km downstream of landfill | 2 |
| No designated site within 250 m upstream or 1 km downstream of landfill | 1 |

Ideally, the proximity of historic coastal landfills to other areas known to be suitable habitat for protected species or that contain protected species would also be assessed. The interactive maps on MAGIC (Defra, 2013b) could be used for this purpose, but landfill site locations cannot be overlaid and comparing their locations would be extremely time consuming due to the large number of ecological sites and landfills to consider. In addition, local knowledge also needs to be applied as not all sites or species, e.g. Local Wildlife Sites and herptiles, are included on the MAGIC maps. The National Biodiversity Network (NBN) website (https://data.nbn.org.uk/Datasets) allows searching of its database for species records within a 10 km scale, but the size of the area reported for each search and the way the data are presented, requires checking hundreds of individual links for each landfill site, and makes this tool impracticable for the purposes of this research. Therefore, it would be more appropriate to undertake this level of assessment on the individual landfill sites that are found to pose the highest risk based on other factors during the screening.

#### 5.5.2.3 Controlled waters – water quality

The current water quality of the receiving waterbody will be a factor in its vulnerability as it will determine its capacity to buffer any input of contaminated leachate. Environment Agency maps showing the Water Framework Directive ecological status (high, good, moderate, poor or bad) and chemical status of surface waters (good or failing to achieve good) in England can be downloaded from the Defra website in pdf format (Environment Agency, 2015c; 2015d), but the small scale of these maps makes it impracticable to compare landfill locations to them. In addition, it would be necessary to know whether the input of contaminants would cause a change in status and there is insufficient information available about contaminant loads to be able to determine this.

However, WFD classifications are based on annual average chemical concentrations in watercourses and it is likely that metals released from the eroded waste will only have a short residence time. Metals in sediment rapidly desorb in saline water, i.e. in minutes to hours (Millward and Liu, 2003), concentrations of released metals are likely to be very low (see Chapter 4), and breaching of a site is more likely to occur during extreme events such as flooding or tidal surges when the dilution and water exchange will be even greater than normal. Hence, metal concentrations in water are unlikely to have any impact upon surface water quality. Although it is not likely that metals released from eroded waste will affect surface water quality, other contaminants may pose a risk and it is possible that if large volumes of organic matter are released deoxygenation of the water around the landfill may occur. The impact of these will depend upon dilution in the receiving waters, which has already been accounted for in the pollution index as part of the landfill hazard (see section 5.4.2.3) and, therefore, was not considered here.

Surface water Drinking Water Safeguard Zones and locations with surface water Water Abstraction Licences can be viewed on the Environment Agency website (Environment Agency, 2016b; 2016f). As historic landfill locations cannot be overlaid onto the maps and there is no accurate scale (the scale is given as a ratio, but does not define the screen size the ratio is accurate for), the proximity of these sites cannot be easily determined and considering them would be more appropriate for detailed investigations of the sites determined to pose the highest pollution risk based on other factors. Therefore, these zones are not considered in the screening assessment.

#### 5.5.2.4 Property

**Crops and animals that may be consumed by humans**

Seaweed, crustaceans, other shellfish and fish may be harvested from the intertidal zone and tidal waters for human consumption. Only GIS datasets relating to shellfish waters could be found: the Classified Bivalve Mollusc Harvesting Areas GIS dataset is available upon request from Cefas (O. Morgan, pers. comm., email, 2/11/2015) and the Shellfish Waters GIS dataset can be downloaded from Defra's MAGIC website (Defra, 2016). These include habitats of gastropod and bivalve molluscs, but not shellfish crustaceans, e.g. crayfish, crabs and lobsters (Defra, 2011). Distances between these areas and historic coastal landfills can easily be assessed on a national scale using GIS mapping software if distance ranges (buffer zones) are used rather than absolute

distances to each site. Therefore, similar to assessing the vulnerability of human receptors, distances between these areas and historic coastal landfill sites were determined using GIS software and used as a proxy for the quantities of solid waste materials and dissolved contaminants that may reach these areas. This was based on the assumption that the greater the distance from the source of the waste, the greater the dispersion of the waste and dilution of the contaminants. The seafood severity scores used are shown in Table 5.27. Sites upstream of the landfills are included to account for tidal movement of contaminants. The upper buffer limit of 1 km was selected for the same reasons as it was used for the designated sites (see section 5.5.2.2).

**Table 5.27: Determining the seafood severity score**

| Distance between landfill site and Shellfish Waters or Classified Bivalve Mollusc Harvest Areas | Seafood severity score |
|---|---|
| Landfill is within one or more shellfish/mollusc site(s) | 5 |
| Shellfish/mollusc site within 100 m upstream or downstream of landfill | 4 |
| Shellfish/mollusc site within 250 m upstream or 500 m downstream of landfill | 3 |
| Shellfish/mollusc site within 250 m upstream or 1 km downstream of landfill | 2 |
| No shellfish/mollusc site within 250 m upstream or 1 km downstream of landfill | 1 |

### 5.5.3 Calculation of the environmental vulnerability sub-index

The selected parameters for the environmental vulnerability sub-index are summarised in Table 5.28. When summed, the parameters selected result in an environmental vulnerability sub-index value ranging from 3 to 15. Therefore, the formula to normalise this score to a percentage to enable the sub-indices to be combined into the overall risk index (after McLaughlin and Cooper, 2010) is as shown in Equation 5.10.

**Table 5.28: Summary of environmental vulnerability sub-index parameters severity scoring**

| Parameters | Severity scores |
|---|---|
| Human impact | Distance to bathing water catchment: 1: distance > 150 m<br>2: 100 m < distance ≤ 150 m<br>3: 50 m < distance ≤ 100 m<br>4: 0 m < distance ≤ 50 m<br>5: Landfill site is in bathing water catchment |
| Designated sites | 1: No designated site within 250 m upstream or 1 km downstream<br>2: Designated site within 250 m upstream or 1 km downstream<br>3: Designated site within 250 m upstream or 500 m downstream<br>4: Designated site within 100 m upstream or downstream<br>5: Landfill site is within one or more designated site(s) |
| Seafood | 1: No shellfish/mollusc site within 250 m upstream or 1 km downstream of landfill<br>2: Shellfish/mollusc site within 250 m upstream or 1 km downstream of landfill<br>3: Shellfish/mollusc site within 250 m upstream or 500 m downstream of landfill<br>4: Shellfish/mollusc site within 100 m upstream or downstream of landfill<br>5: Landfill is within one or more shellfish/mollusc site(s) |
| **Minimum environmental vulnerability sub-index value = 3**<br>**Maximum environmental vulnerability sub-index value = 15** | |

**Equation 5.10: Calculation of the normalised environmental vulnerability sub-index**

$$Normalised\ environmental\ vulnerability\ sub\text{-}index = \frac{\sum severity\ scores - 3}{12}\ x\ 100$$

## 5.6 Summary of the screening method

To aid understanding of the overall assessment process the flow of data from the parameters to the overall risk is summarised in Figure 5.5.

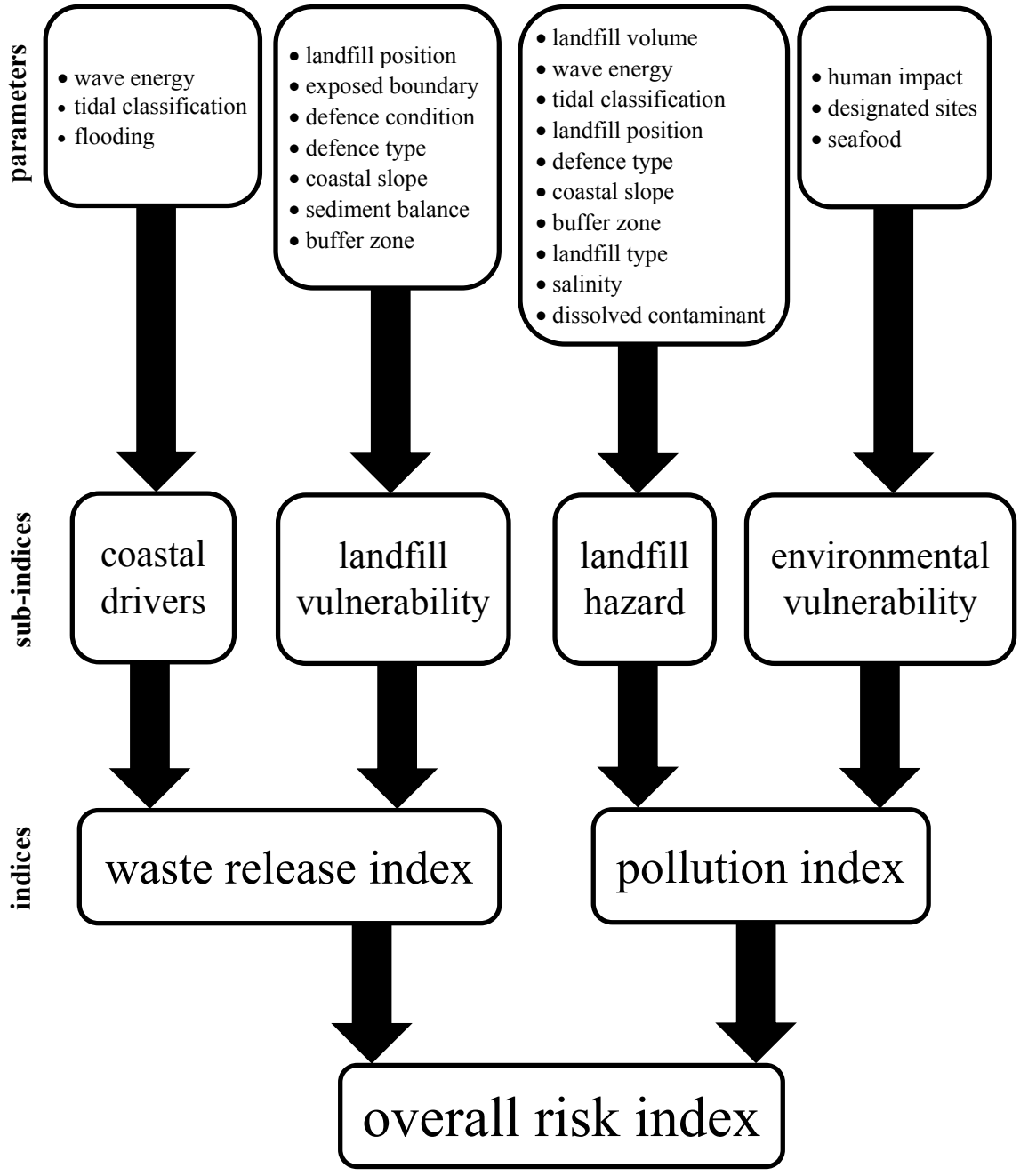



**Figure 5.5: Flow chart showing the relationship between the parameters, sub-indices, indices and overall risk index**

## 5.7 Sensitivity analysis of the risk screening assessment

A range sensitivity method has been applied to the risk screening assessment to determine which parameters have the greatest influence on the indices values (United States Environmental Protection Agency, 2001). The sensitivity ratio was determined by varying the severity score for each parameter in turn from the baseline value of 3, the mid-point on the severity scale, to the maximum possible severity score of 5 and substituting the resulting index values into Equation 5.11 (United States Environmental Protection Agency, 2001; Saltelli et al., 2008). The sensitivity ratio was determined for the waste release index, pollution index and overall risk index, the results can be seen in Table 5.29. The overall risk index is most sensitive to variations in the wave energy and tidal classification severity scores and least sensitivity to changes the landfill volume, landfill type, salinity and dissolved contaminant severity scores.

**Equation 5.11: Calculation of the sensitivity ratio (after United States Environmental Protection Agency, 2001)**

$$Sensitivity\ ratio = \frac{\frac{Y_2 - Y_1}{Y_1}\ x\ 100\%}{\frac{X_2 - X_1}{X_1}\ x\ 100\%}$$

*Where:*
*$Y_1$ = the baseline index value (i.e. when all severity scores = 3)*
*$Y_2$ = the index value when the parameter's severity score is $X_2$*
*$X_1$ = the baseline severity score for the parameter being tested*
*$X_2$ = the maximum possible severity score for the parameter being tested*

**Table 5.29: Results of the Range Sensitivity Ratio analyses (from highest to lowest sensitivity ratio for overall risk)**

| | Range Sensitivity Ratio for the: | | |
|---|---|---|---|
| **Parameter** | **Waste release index** | **Pollution index** | **Overall risk index** |
| Wave energy | 0.250 | 0.075 | 0.163 |
| Tidal classification | 0.250 | 0.075 | 0.163 |
| Flooding | 0.250 | 0.000 | 0.125 |
| Human impact | 0.000 | 0.250 | 0.125 |
| Designated sites | 0.000 | 0.250 | 0.125 |
| Seafood | 0.000 | 0.250 | 0.125 |
| Landfill position | 0.107 | 0.075 | 0.091 |
| Defence type | 0.107 | 0.075 | 0.091 |
| Coastal slope | 0.107 | 0.075 | 0.091 |
| Buffer zone | 0.107 | 0.075 | 0.091 |
| Exposed boundary length | 0.107 | 0.000 | 0.054 |
| Defence condition | 0.107 | 0.000 | 0.054 |
| Sediment balance | 0.107 | 0.000 | 0.054 |
| Landfill volume | 0.000 | 0.075 | 0.038 |
| Landfill type | 0.000 | 0.075 | 0.038 |
| Salinity | 0.000 | 0.075 | 0.038 |
| Dissolved contaminant | 0.000 | 0.075 | 0.038 |

## 5.8 Sites selected for testing the screening assessment

In Essex there are 157 historic coastal landfills with unique Historic Landfill Database reference numbers recorded in the EA's database (Environment Agency, 2015b). Eight of these were selected for testing the screening assessment methodology: Common Road in Great Wakering, Hadleigh Marsh in Hadleigh, Leigh Marshes in Leigh on sea, Martins Farm South and Martins Farm North in St Osyth, Newlands on Canvey Island, Park Drive in Maldon, and Sea Wall in South Fambridge (Table 5.30). These sites were selected because access to site records was readily available as they are managed by Essex County Council and Southend Borough Council who were partners in the funding of this research. The landfills are distributed over four estuaries allowing the method to be tested in different environments, but it is also important that the method can differentiate the pollution risk from adjacent sites to support the local allocation of remediation funds. Therefore, some of the landfills are adjacent to each other, which allows testing of the method for sensitivity to changes in factors such as the distance between the landfill and mean high water. A summary of the sites' histories is shown in Table 5.30. Figure 5.6 shows the relative locations of the selected sites around the coastline of Essex, and Figure 5.7 shows the surface areas of the landfills to scale.

**Table 5.30: Screening assessment test site histories**

| Name and landfill database reference no.[1] | Operating period[2] | Type[3] | Volume[2] ($m^3$) | Flood defences[1] |
|---|---|---|---|---|
| **Common Road** EAHLD01226 | 1970-1993 | Household, commercial and industrial | 450,000 | Partly defended |
| **Hadleigh Marsh** EAHLD01181 | 1980-1987 | Household and commercial | 500,000 | Landfill is the flood defence |
| **Leigh Marshes** EAHLD00531 | 1955-1967 | Household, commercial and industrial | 800,000 | Yes |
| **Martins Farm North** EAHLD01246 | 1960-1995 | Household, commercial and industrial | 1,400,000 | Yes |
| **Martins Farm South** EAHLD01241 | 1985-1995 | Household, commercial and industrial | 1,200,000 | Yes |
| **Newlands** EAHLD01178 | 1954-1989 | Household, commercial and industrial | 1,000,000 | Landfill is the flood defence |
| **Park Drive** EAHLD01739 | 1974-1994 | Household, commercial and industrial | 800,000 | Landfill is the flood defence |
| **Sea Wall** EAHLD01228 | 1988-1991 | Household, commercial and industrial | 275,000 | Landfill is the flood defence |

[1] GIS datasets from UK Government (2016)
[2] Site records (A. Brown, pers. comm., email, 26/10/2015), except Leigh Marshes operating period from Environment Agency (2013f) and volume estimated using GIS data from UK Government (2016) and trial pit depths recorded in a report by Halcrow (2012).
[3] Environment Agency (2013f)


**Figure 5.6: Map showing an overview of the locations of the historic coastal landfill sites selected for testing the risk screening assessment (Contains Ordnance Survey data © Crown copyright and database right 2016. © Crown copyright. All rights reserved). (North up the page)**

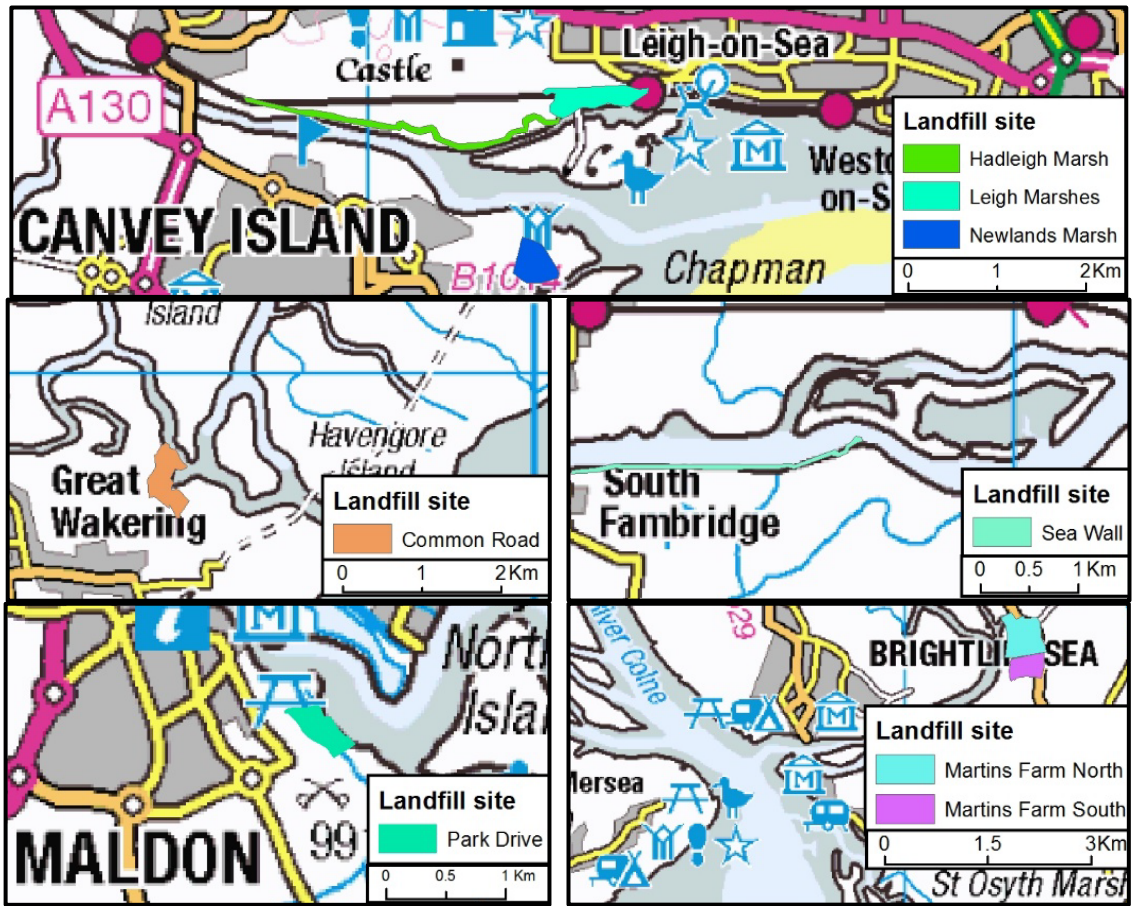



**Figure 5.7: Maps showing the areas of the landfills used for testing the risk screening assessment (Contains Ordnance Survey data © Crown copyright and database right 2016. © Crown copyright. All rights reserved). (North up the page)**

## 5.9 Worked example of the risk screening assessment calculations

As a demonstration of calculating the sub-indices and indices using the gathered severity score data, a worked example is shown here using data for Hadleigh Marsh waste filled flood embankment.

The severity scores determined for the coastal drivers parameters for Hadleigh Marsh waste filled embankment are shown in Table 5.31. To determine the coastal drivers sub-index the coastal drivers parameters' severity scores were summed, giving a coastal drivers sub-index = 4. The coastal drivers sub-index was then normalised using Equation 5.12, which gave a normalised coastal drivers sub-index = 8.3.

**Table 5.31: Coastal drivers sub-index data for a worked example of the risk screening assessment calculations using Hadleigh Marsh as a test site**

| Parameters for the coastal drivers sub-index | Severity scores |
|---|---|
| Wave energy | 1 |
| Tidal classification | 1 |
| Flooding | 2 |
| *Sum of severity scores* | *4* |

**Equation 5.12: Calculation of the normalised coastal drivers sub-index for Hadleigh Marsh**

$$Normalised\ coastal\ driver\ sub\text{-}index = \frac{4-3}{12}\ x\ 100 = 8.3$$

The severity scores determined for the landfill vulnerability parameters for Hadleigh Marsh waste filled flood embankment are shown in Table 5.32. To determine the landfill vulnerability sub-index the landfill vulnerability parameters' severity scores were summed, giving a landfill vulnerability sub-index = 31. The landfill vulnerability sub-index was then normalised using Equation 5.13, which gave a normalised landfill vulnerability sub-index = 85.7.

**Table 5.32: Landfill vulnerability sub-index data for a worked example of the risk screening assessment calculations using Hadleigh Marsh as a test site**

| Parameters for the landfill vulnerability sub-index | |
|---|---|
| Landfill position | 5 |
| Exposed boundary length | 5 |
| Defence condition | 3 |
| Defence type | 5 |
| Coastal slope | 4 |
| Sediment balance | 5 |
| Buffer zone | 4 |
| *Sum of severity scores* | *31* |

**Equation 5.13: Calculation of the normalised landfill vulnerability sub-index for Hadleigh Marsh**

$$Normalised\ landfill\ vulnerability\ sub\text{-}index = \frac{31-7}{28}\ x\ 100 = 85.7$$

The severity scores determined for the landfill hazard parameters for Hadleigh Marsh waste filled flood embankment are shown in Table 5.33. To determine the landfill hazard sub-index the landfill hazard parameters' severity scores were summed, giving a landfill hazard sub-index = 28. The landfill hazard sub-index was then normalised using Equation 5.14, which gave a normalised landfill hazard sub-index = 45.0.

**Table 5.33: Landfill hazard sub-index data for a worked example of the risk screening assessment calculations using Hadleigh Marsh as a test site**

| Parameters for the landfill hazard sub-index | |
|---|---|
| Landfill volume | 1 |
| Wave energy | 1 |
| Tidal classification | 1 |
| Landfill position | 5 |
| Defence type | 5 |
| Coastal slope | 4 |
| Buffer zone | 4 |
| Landfill type | 2 |
| Salinity | 4 |
| Dissolved contaminant | 1 |
| *Sum of severity scores* | *28* |

**Equation 5.14: Calculation of the normalised landfill hazard sub-index for Hadleigh Marsh**

$$Normalised\ landfill\ hazard\ sub\text{-}index = \frac{28-10}{40}\ x\ 100 = \ 45.0$$

The severity scores determined for the environmental vulnerability parameters for Hadleigh Marsh waste filled flood embankment are shown in Table 5.34. To determine the environmental vulnerability sub-index the environmental vulnerability parameters' severity scores were summed, giving an environmental vulnerability sub-index = 12. The environmental vulnerability sub-index was then normalised using Equation 5.15, which gave a normalised environmental vulnerability sub-index = 75.0.

**Table 5.34: Environmental vulnerability sub-index data for a worked example of the risk screening assessment calculations using Hadleigh Marsh as a test site**

| **Parameters for the environmental vulnerability sub-index** | |
|---|---|
| Human impact | 5 |
| Designated sites | 5 |
| Seafood | 2 |
| *Sum of severity scores* | *12* |

**Equation 5.15: Calculation of the normalised environmental vulnerability sub-index for Hadleigh Marsh**

$$Normalised\ environmental\ vulnerability\ sub\text{-}index = \frac{12-3}{12}\ x\ 100 = \ 75.0$$

To calculate the waste release index, the values of the normalised coastal drivers sub-index and normalised landfill vulnerability sub-index were substituted into Equation 5.16, which gave a waste release index = 47.0.

**Equation 5.16: Calculation of the waste release index**

$$Waste\ release\ index = \frac{normalised\ coastal\ drivers + normalised\ landfill\ vulnerability}{2}$$

$$= \frac{8.3 + 85.7}{2} = \ 47.0$$

To calculate the pollution index, the values of the normalised landfill hazard and normalised environmental vulnerability were substituted into Equation 5.17, which gave a waste release index = 60.0.

**Equation 5.17: Calculation of the pollution index for Hadleigh Marsh**

$$Pollution\ index = \frac{normalised\ landfill\ hazard + normalised\ environmental\ vulnerability}{2}$$

$$= \frac{45.0 + 75.0}{2} = 60.0$$

Finally, to calculate the overall risk, the values of the waste release index and pollution index were substituted into Equation 5.18, which gave an overall risk index = 53.5.

**Equation 5.18: Calculation of the overall risk index for Hadleigh Marsh**

$$Overall\ risk\ index = \frac{waste\ release\ index + pollution\ index}{2} = \frac{47.0 + 60.0}{2} = 53.5$$

## 5.10 Results of testing the screening method

### 5.10.1 Assessment of the risk of waste being released

#### *Coastal drivers sub-index*

The severity scores for the coastal drivers sub-index parameters for the test sites are shown in Table 5.35. The range of possible coastal driver sub-index values is 3 to 15. Leigh Marshes and Sea Wall historic coastal landfills had the joint highest coastal drivers sub-index value of the sites tested, scoring 5; all other sites achieved the lowest possible coastal drivers sub-index value of 3, with the exception of Hadleigh Marsh which scored 4.

**Table 5.35: Severity scores for the coastal drivers sub-index parameters for the test sites, sites shown left to right from highest to lowest coastal drivers sub-index value**

| Parameter being scored | Leigh Marshes | Sea Wall | Hadleigh Marsh | Common Road | Martins Farm North | Martins Farm South | Newlands | Park Drive |
|---|---|---|---|---|---|---|---|---|
| **Wave energy** | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| **Tidal classification** | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| **Flooding** | 3 | 3 | 2 | 1 | 1 | 1 | 1 | 1 |
| **Coastal drivers sub-index value** | **5** | **5** | **4** | **3** | **3** | **3** | **3** | **3** |

#### *Landfill vulnerability sub-index*

The severity scores for the landfill vulnerability sub-index parameters for the test sites are shown in Table 5.36. The range of possible landfill vulnerability sub-index values is 7 to 35. Hadleigh Marsh had the highest landfill vulnerability sub-index value of the sites tested, scoring 31, and Martins Farm South had the lowest landfill vulnerability sub-index value of the sites tested, scoring 20.

**Table 5.36: Severity scores for the landfill vulnerability sub-index parameters for the test sites, sites shown left to right from highest to lowest landfill vulnerability sub-index value**

| Parameter being scored | Hadleigh Marsh | Sea Wall | Newlands | Leigh Marshes | Common Road | Martins Farm North | Park Drive | Martins Farm South |
|---|---|---|---|---|---|---|---|---|
| **Landfill position** | 5 | 5 | 5 | 5 | 5 | 4 | 4 | 1 |
| **Exposed boundary length** | 5 | 5 | 3 | 3 | 3 | 2 | 1 | 2 |
| **Defence condition** | 3 | 3 | 3 | 3 | 3 | 4 | 2 | 3 |
| **Defence type** | 5 | 5 | 5 | 2 | 4 | 3 | 5 | 3 |
| **Coastal slope** | 4 | 1 | 5 | 4 | 1 | 1 | 5 | 1 |
| **Sediment balance** | 5 | 5 | 5 | 3 | 3 | 5 | 3 | 5 |
| **Buffer zone** | 4 | 5 | 1 | 5 | 5 | 5 | 4 | 5 |
| **Landfill vulnerability sub-index value** | **31** | **29** | **27** | **25** | **24** | **24** | **24** | **20** |

## *Waste release index*

The normalised sub-indices values and waste release index calculated for each of the test sites is shown in Table 5.37. The range of possible waste release index values is 0 to 100. Of the sites tested, Sea Wall in South Fambridge had the highest waste release index, with a value of 47.6, and Martins Farm South had the lowest waste release index, with a value of 23.2.

**Table 5.37: Test sites ranked from highest to lowest waste release index value**

| Landfill name | Normalised sub-indices values | | Waste release index |
|---|---|---|---|
| | Coastal drivers | Landfill vulnerability | |
| **Sea Wall** | 16.7 | 78.6 | **47.6** |
| **Hadleigh Marsh** | 8.3 | 85.7 | **47.0** |
| **Leigh Marshes** | 16.7 | 64.3 | **40.5** |
| **Newlands** | 0.0 | 71.4 | **35.7** |
| **Common Road** | 0.0 | 60.7 | **30.4** |
| **Martins Farm North** | 0.0 | 60.7 | **30.4** |
| **Park Drive** | 0.0 | 60.7 | **30.4** |
| **Martins Farm South** | 0.0 | 46.4 | **23.2** |

### 5.10.2 Assessment of the risk to receptors from released waste

#### *Landfill hazard sub-index*

The severity scores for the landfill hazard sub-index parameters for the test sites are shown in Table 5.38. The range of possible landfill hazard sub-index values is 10 to 50. Parks Drive had the highest landfill hazard sub-index value of the sites tested, scoring 31, and Martins Farm South had the lowest landfill hazard sub-index value of the sites tested, scoring 25.

**Table 5.38: Severity scores for the landfill hazard sub-index parameters for the test sites, sites shown left to right from highest to lowest landfill hazard sub-index value**

| Parameter being scored | Park Drive | Sea Wall | Martins Farm North | Hadleigh Marsh | Leigh Marshes | Newlands | Common Road | Martins Farm South |
|---|---|---|---|---|---|---|---|---|
| **Landfill volume** | 2 | 1 | 3 | 1 | 2 | 2 | 1 | 3 |
| **Wave energy** | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| **Tidal classification** | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| **Landfill position** | 4 | 5 | 4 | 5 | 5 | 5 | 5 | 1 |
| **Defence type** | 5 | 5 | 3 | 5 | 2 | 5 | 4 | 3 |
| **Coastal slope** | 5 | 1 | 1 | 4 | 4 | 5 | 1 | 1 |
| **Buffer zone** | 4 | 5 | 5 | 4 | 5 | 1 | 5 | 5 |
| **Landfill type** | 3 | 3 | 3 | 2 | 3 | 3 | 3 | 3 |
| **Salinity** | 4 | 5 | 3 | 4 | 4 | 4 | 5 | 3 |
| **Dissolved contaminant** | 2 | 2 | 4 | 1 | 1 | 1 | 2 | 4 |
| **Landfill hazard sub-index value** | **31** | **29** | **28** | **28** | **28** | **28** | **28** | **25** |

#### *Environmental vulnerability sub-index*

The severity scores for the environmental vulnerability sub-index parameters for the test sites are shown in Table 5.39. The range of possible environmental vulnerability sub-index values is 3 to 15. Common Road had the highest environmental vulnerability sub-index value of the sites tested, achieving the maximum possible score of 15, and Park Drive and Sea Wall had the lowest environmental vulnerability sub-index values of the sites tested, both scoring 11.

**Table 5.39: Severity scores for the environmental vulnerability sub-index parameters for the test sites, sites shown left to right from highest to lowest environmental vulnerability sub-index value**

| Parameter being scored | Common Road | Martins Farm North | Martins Farm South | Newlands | Hadleigh Marsh | Leigh Marshes | Park Drive | Sea Wall |
|---|---|---|---|---|---|---|---|---|
| **Human impact** | 5 | 5 | 5 | 5 | 5 | 5 | 5 | 1 |
| **Designated sites** | 5 | 5 | 5 | 5 | 5 | 5 | 5 | 5 |
| **Seafood** | 5 | 4 | 3 | 3 | 2 | 2 | 1 | 5 |
| **Environmental vulnerability sub-index value** | **15** | **14** | **13** | **13** | **12** | **12** | **11** | **11** |

### *Pollution index*

The normalised sub-indices scores and pollution index calculated for each of the test sites is shown in Table 5.40. The range of possible pollution index values is 0 to 100. Of the sites tested, Common Road had the highest pollution index, with a value of 72.5, and Sea Wall in South Fambridge had the lowest pollution index, with a value of 57.1.

**Table 5.40: Test sites ranked from highest to lowest pollution index value**

| Landfill name | Normalised sub-indices values | | Pollution index |
|---|---|---|---|
| | Landfill hazard | Environmental vulnerability | |
| **Common Road** | 45.0 | 100.0 | **72.5** |
| **Martins Farm North** | 45.0 | 91.7 | **68.3** |
| **Newlands** | 45.0 | 83.3 | **64.2** |
| **Martins Farm South** | 37.5 | 83.3 | **60.4** |
| **Leigh Marshes** | 45.0 | 75.0 | **60.0** |
| **Hadleigh Marsh** | 45.0 | 75.0 | **60.0** |
| **Park Drive** | 52.5 | 66.7 | **59.6** |
| **Sea Wall** | 47.5 | 66.7 | **57.1** |

## 5.10.3 Overall risk index

The overall risk index calculated for each of the test sites is shown in Table 5.41. The range of possible overall risk index values is 0 to 100. Of the sites tested, Hadleigh Marsh had the highest overall risk index, with a value of 53.5, and Martins Farm South had the lowest overall risk index, with a value of 41.8.

**Table 5.41: Test sites ranked by overall risk index from highest risk to lowest risk**

| Landfill name | Waste release index | Pollution index | Overall risk index |
|---|---|---|---|
| **Hadleigh Marsh** | 47.0 | 60.0 | 53.5 |
| **Sea Wall** | 47.6 | 57.1 | 52.4 |
| **Common Road** | 30.4 | 72.5 | 51.4 |
| **Leigh Marshes** | 40.5 | 60.0 | 50.2 |
| **Newlands** | 35.7 | 64.2 | 49.9 |
| **Martins Farm North** | 30.4 | 68.3 | 49.3 |
| **Park Drive** | 30.4 | 59.6 | 45.0 |
| **Martins Farm South** | 23.2 | 60.4 | 41.8 |

## 5.11 Discussion

Obtaining the necessary data and applying the risk screening assessment to the test sites proved to be straightforward, although additional categories had to be added to the defence type severity scoring. This was to include additional defence type options that were not initially considered, specifically the possibility of defences being a mixture of hard and soft, and the possibility that the landfill is only partially defended. The boundaries of sites were often found to overlap, which appears to be related to changes in license holders or licensing conditions, e.g. permitted waste types, and some sites were represented by more than one polygon when viewed in GIS software, e.g. ArcMap. Sites are assigned unique Historic Landfill Database reference numbers, referred to as hld_ref, in the GIS datasets which were used to distinguish sites for assessment. Some errors were found in the site boundaries, e.g. the GIS landfill boundaries show a strip of landfill between Leigh Marshes and Two Tree Island landfills in Essex where in reality there is an open creek with a channel that is deeper than the waste in the two landfill sites. Therefore, it was necessary to compare the GIS dataset to background maps to check for anomalies and use personal judgement in determining the correct boundary to use for the assessment. Issues such as these could be resolved during the site verification step (step 2) of a three step assessment.

The coastal drivers sub-index ranked Leigh Marshes and Sea Wall in South Fambridge as the sites potentially subjected to the greatest drivers of erosion, followed by Hadleigh Marsh. However, the landfill vulnerability sub-index indicated that Leigh Marshes is better protected from the coastal drivers than Sea Wall and Hadleigh Marsh, which reflects the fact that it has a much shorter length of boundary facing mean high water and is separated from the estuary by a flood defence. In contrast, Sea Wall and Hadleigh Marsh are both waste filled flood embankments with several kilometres of exposed boundary. The waste release index, which combines the coastal drivers and landfill vulnerability sub-indices, indicated the two flood embankments (Sea Wall in South Fambridge and Hadleigh Marsh) are the two most likely test sites to release solid waste to the environment, reflecting their exposure to their estuaries, having no flood defences separating them from the water and having very long boundaries adjacent to mean high water, increasing the probability that at least part of the landfill sites will breach.

In contrast, the two waste filled flood embankments were ranked low in the range of pollution index values suggesting that, if waste erodes from them, they are likely to cause comparatively less pollution than the other sites tested. This reflects the relatively small volumes of waste in the two flood embankments, combined with the high levels of dilution at Hadleigh Marsh landfill site and the absence of bathing water catchments in the estuary at the Sea Wall in South Fambridge landfill site. However, the two waste filled flood embankments had the two highest overall risk index values reflecting that, for the test sites, the range of waste release index values (range = 24.4) is greater than the range of pollution index values (range = 15.4) and therefore the waste release index has greater influence in determining the overall risk index ranking of the test sites. The limited range of pollution index values reflects the very similar waste contents and ecological environments of the eight sites, and the greater range of waste release index values reflects the greater range of vulnerabilities of the landfill sites to coastal drivers, particularly differences in defences and the lengths of their boundaries.

For the landfill hazard sub-index, 5 of the 8 sites had the same value (28), but achieved the value for different reasons. This highlights that it is important that the method used to determine the sub-indices and indices is reported, and that a record is kept of the severity scores for individual parameters to aid understanding of which can be addressed to reduce risk at sites with a high overall risk.

For the test sites, 7 of the 8 contained the same types of waste (industrial, commercial and household), but a greater range of landfill type severity scores will be achieved if the method is used for a national assessment. Of the 1264 known historic coastal landfills, 466 contain only one waste type, 572 contain more than one type, 23 sites are specifically categorised as waste type unknown, and an additional 203 do not have a waste type listed.

Sensitivity analysis showed the pollution index is more sensitive to changes in the environmental vulnerability sub-index parameters than the landfill hazard sub-index parameters. It could be argued that the vulnerability of receptors is more important in determining pollution risk than the chemical content of the material released from an eroding landfill sites as it is likely that all landfilled materials have the potential to be ecologically harmful in some manner, for example smothering the intertidal zone with their presence in large quantities or harming fauna by being mistaken for food (plastic

waste). However, whether the vulnerability of the receptors is more important in determining pollution risk than the volume of the waste released is not as clear and needs further investigation. This highlights that further consideration needs to be given to which of the parameters, if any, are more significant than others in determining the overall risk in reality and, hence, whether weightings should be applied to them to increase their influence on the final risk rankings. Some researchers have chosen not to weight parameters in their risk assessment methods due to the subjectivity of such weightings (e.g. McLaughlin and Cooper, 2010), whilst others have weighted individual parameters based on surveys of industry and academic experts (e.g. Kumar and Alappat, 2005; Sharma et al., 2008; Singh et al., 2009). Weightings from previous risk assessments cannot be adopted for this risk assessment as they have used different parameters and indices, and weightings must be specific to the combination of parameters and indices being used. Therefore, to determine weightings with any useful level of accuracy would require input from experts in coastal processes, landfill engineering stability and contamination, and ecology. The necessary consultations were beyond the scope of this research, but should be considered in any future developments of the risk screening assessment.

The use of the transitional area of the estuary from the WFD transitional and coastal waterbodies cycle 2 dataset to calculate the tidal prism potentially overestimates the dilution of contaminants for landfill sites that are on tributaries that are not considered independently in the WFD dataset, e.g. Martins Farm North and South (see Figure 5.8), Sea Wall in South Fambridge and Common Road. This level of accuracy in determining dilution was considered appropriate given the uncertainty associated with the concentrations of contaminants in the waste and their mobility. Testing the sensitivity of the indices scores for Martins Farm North and South, Sea Wall and Common Road by only including the area of the tributary they are on in the tidal prism calculation, rather than the entire transitional area, found there were no effects on the indices scores for Martins Farm North or South. However, for Sea Wall and Common Road the pollution indices increased by 1.25 and the overall risk indices by 0.625. This did not affect the overall risk rankings of the test sites.

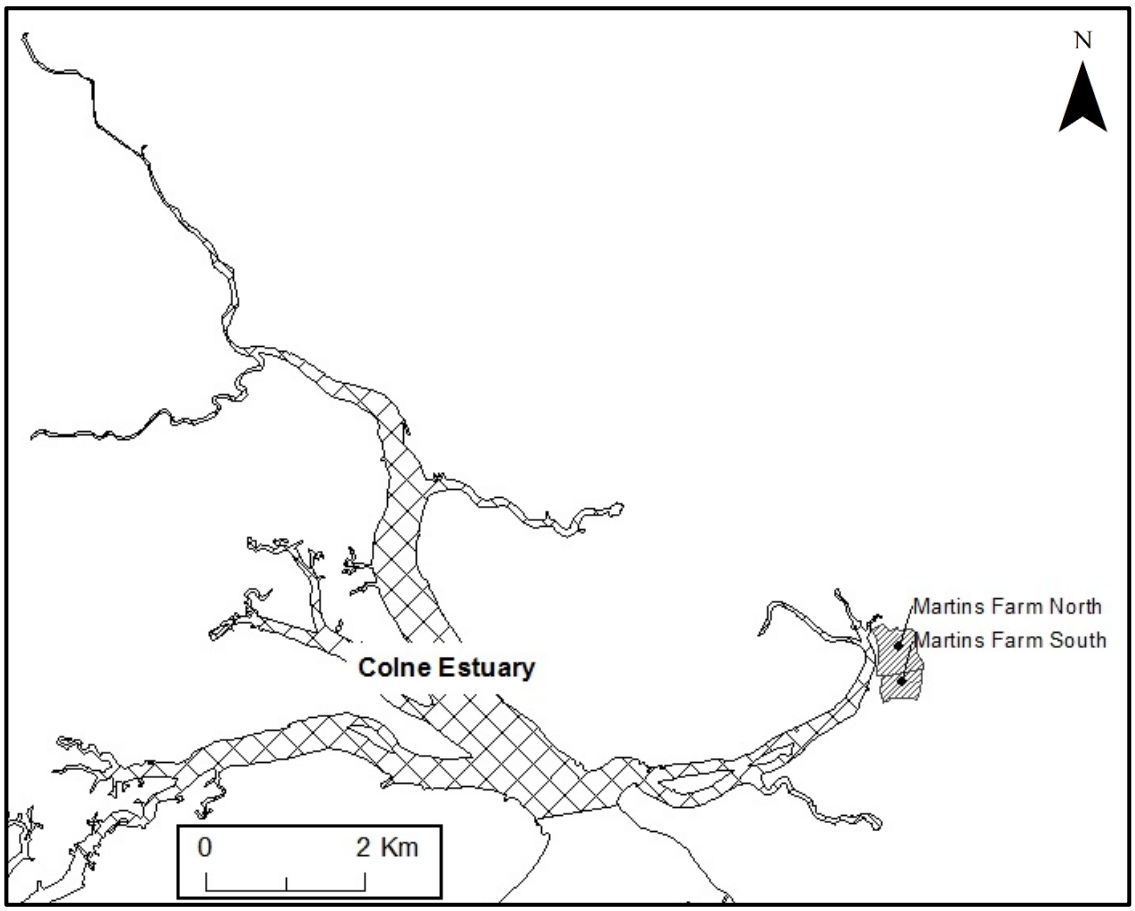



**Figure 5.8: The Colne Estuary transitional area in the WFD transitional and coastal waterbodies cycle 2 dataset includes multiple tributaries, which could result in the tidal prism volume for diluting contaminants from Martins Farm North and South landfill sites being overestimated (Contains Environment Agency information © Environment Agency and/or database right)**

The value of the overall risk index can range from 0 to 100 under the proposed scoring system. As there are 1264 (currently known) historic coastal landfills to be ranked, there will be multiple landfill sites with similar overall risk index values. One way to address this would be to amend the range to be 0 to 1000, this could be achieved by amending the equations used for normalising the sub-indices scores to include a multiple of 1000 instead of 100 (see Equation 5.19). This would increase the perceived distinction between scores, aiding interpretation (Singh et al., 2009), but would not affect the ranking of sites or the results of the sensitivity analysis. It would also not address the likelihood that there will be clusters of sites with the same indices values, such as was seen for the waste release index where 3 of the 8 sites had an index value of 30.4, but achieved that value through different combinations of parameter severity scores.

**Equation 5.19: Calculation of the normalised sub-indices if the value range were to be increased from 100 to 1000**

$$Normalised\ sub\text{-}index = \frac{\sum severity\ scores - min.possible\ score}{max.possible\ score - min.possible\ score}\ x\ 1000$$

If a series of overall risk index value thresholds were set to provide categories of risk, e.g. very high, high, moderate, low and very low, then this would mitigate the issue of having multiple sites with the same or similar index values. Note a zero risk category is deliberately not included as there is always a residual risk of a site eroding and causing pollution (Neuhold and Nachtnebel, 2011). A categorical risk approach would also have the advantage of allowing the end-user greater discretion in determining the order in which sites are considered for further investigation and/or remedial action, which could better support management of limited budgets. For example, if all sites in a risk category are given the same priority for remediation, rather than using the overall risk score to rank them individually, it would allow multiple sites with low remediation costs to be addressed instead of a single site within the same category that has a higher overall risk score and a higher remediation cost. However, such categories cannot be implemented until a much greater number of sites have been assessed to provide a benchmark of the levels such fixed thresholds should be set at.

An alternative way to show a clearer distinction between index values is to show graphically how the sub-indices contribute to the index value (after Alaska Department of Environmental Conservation, 2015). Figure 5.9, Figure 5.10 and Figure 5.11 demonstrate this approach for the waste release, pollution and overall risk indices respectively. The WEAR method (Alaska Department of Environmental Conservation, 2015) used this approach to show an erosion risk index and contamination risk index for Alaskan landfills. In addition, instead of having an overall risk index, the WEAR method determined the highest risk sites by calculating which sites were in the upper quartile of scores for both indices. This was an appropriate method for the WEAR project as it was undertaken by one team, all sites were visited and then risk assessed at the same time, and there was no requirement to continually update the assessment. It is likely that multiple groups would assess sites around the coast of England, and that the assessments would be continually updated as new data become available as the aim of the risk screening assessment is to identify the highest risk sites and then verify data

through site visits and further investigation where necessary. Therefore, to use the upper quartile of scores to determine the highest risk sites for the proposed risk screening assessment would create a continually changing threshold score. A variation on this approach would be to use fixed threshold scores for indices to group sites into risk categories, e.g. very high, high, moderate, low and very low. This would address the issue of clustered overall risk index values and avoid having continuously changing thresholds, but cannot be implemented until a much greater number of sites has been assessed to provide a benchmark of the levels such fixed thresholds should be set at.

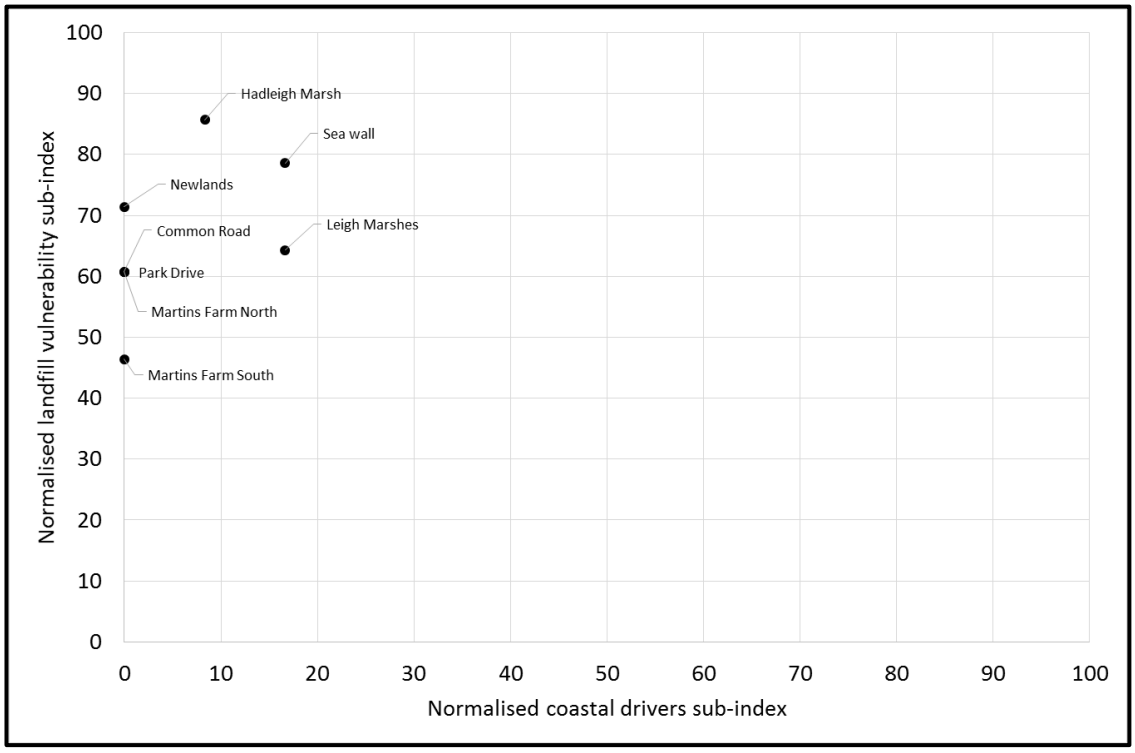



**Figure 5.9: Scatterplot showing the components of the waste release index to aid end-user understanding of the differences between sites**

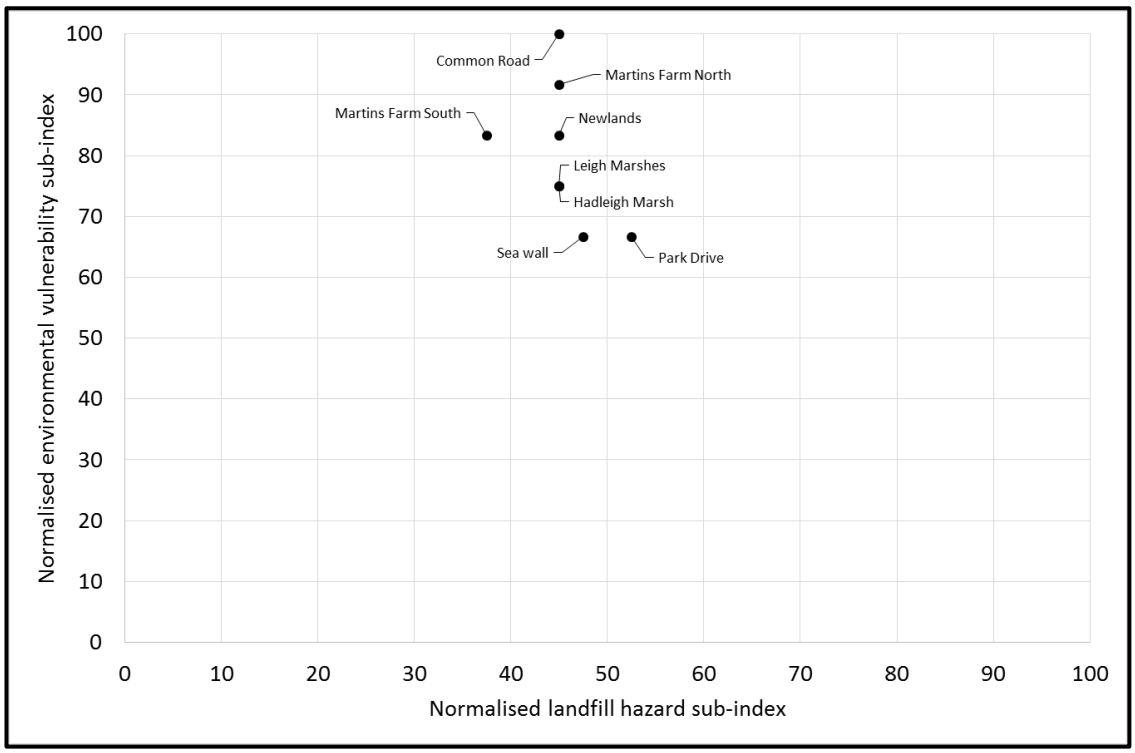



**Figure 5.10: Scatterplot showing the components of the pollution index to aid end-user understanding of the differences between sites**

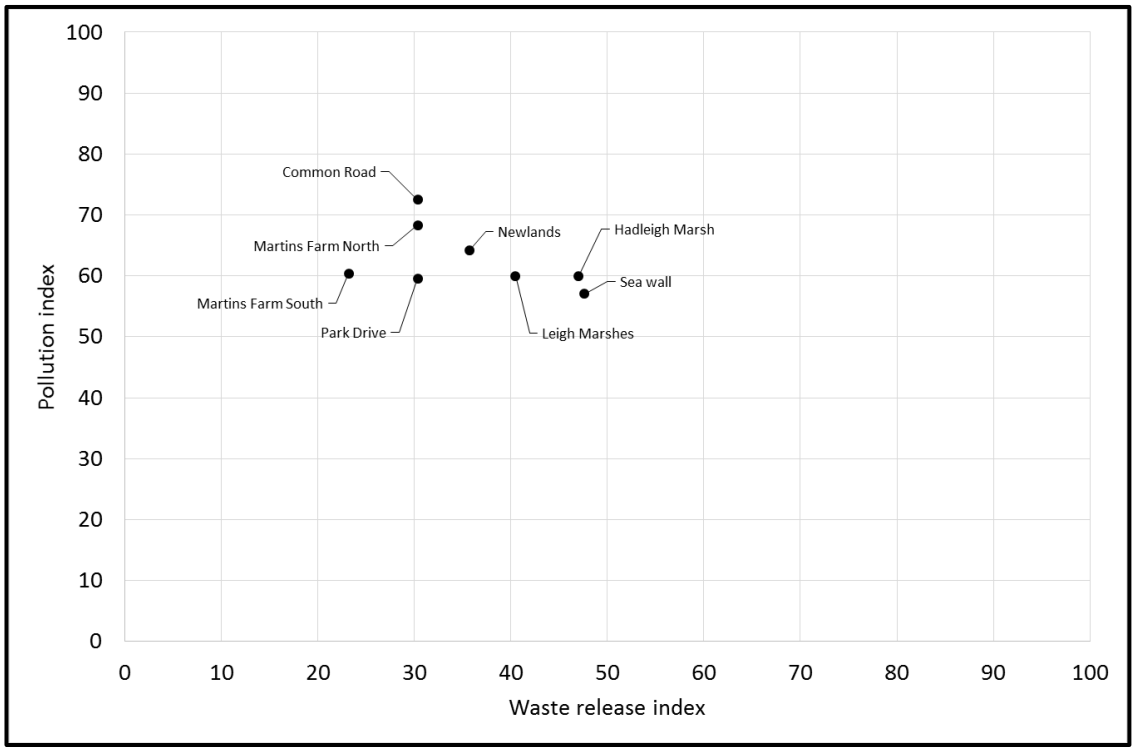



**Figure 5.11: Scatterplot showing the components of the overall risk index to aid end-user understanding of the differences between sites**

### *Future development of the method*

The Coastal Vulnerability Index (CVI) for Great Britain that the British Geological Survey is currently developing should be reviewed, when available, to determine whether it provides an output suitable for use in the risk screening assessment. As the CVI will be a licensable product a cost-benefit analysis of its use should be undertaken if it is suitable for integration into the risk assessment method.

Currently there are no weightings applied to the parameters, sub-indices or indices in the risk assessment. Experts on coastal processes, landfill engineering stability and contamination, and ecology need to be consulted to determine which parameters, sub-indices and indices, if any, are the most important and, consequently, should be weighted to ensure the overall risk index is sufficiently sensitive to them. Such surveys and discussions may also identify additional parameters that have not yet been considered or may identify parameters currently included that are superfluous.

The following potential areas for improvement of the risk screening assessment also require discussion with coastal erosion experts:

1. Further consideration needs to be given to the buffer zone severity score, currently it is based on the average width of saltmarsh, but this may not be appropriate for sites with very long lengths of boundary adjacent to mean high water as some parts of the boundary may be completely unprotected.
2. Further consideration needs to be given to the landfill position severity score, currently it is based on the minimum distance between the landfill boundary and mean high water, but this distance may be reduced to zero by small tributaries that may not have a significant influence on the vulnerability of the site to erosion.
3. Further consideration needs to be given to the flooding severity score. The resolution of the Risk of Flooding from Rivers and Sea (RoFRS) dataset may not be high enough to determine the categories of flood risk for small landfill sites. The RoFRS has well defined external boundaries, but the divides between different flood risk zones are in a much lower resolution and blocky when viewed in GIS software. For small sites it may be more appropriate to use the highest flood risk category the site falls within rather than the flood risk category which covers the greatest proportion of the site.

In addition, the following potential areas for improvement of the risk screening assessment require discussion with regulators:

1. It may be appropriate, subject to approval of the regulators, to include a distinction between different types of ecological sites to ensure that those most difficult to replace or rehabilitate are given priority when considering which landfill sites to remediate the risk from first. For example, protection of designated European Sites, e.g. Ramsars, could potentially receive priority over protection of Local Nature Reserves.
2. The risk screening assessment does not currently consider the cumulative impact of multiple sites eroding. However, where sites are in close proximity, e.g. Hadleigh Marsh and Leigh Marshes landfills, there is the possibility that an event, e.g. storm surge, of sufficient magnitude to breach one site may also breach the other. Consideration needs to be given as to whether sites in close proximity should be subject to a joint assessment as well as individual assessments and, if so, what the minimum separation between sites should be before they are only assessed independently (after UK Government, 2009).

Once the risk screening assessment has been updated with the results of the discussions described above, and a greater number of sites have been assessed, it would be appropriate to use Principal Component Analysis to determine whether the risk screening assessment can be simplified by the removal of any parameters that do not make a significant contribution to the overall risk index (after Pendleton et al., 2010).

When the risk screening assessment has been finalised, it would be advantageous to ensure data are recorded consistently by multiple users and a standardised results format is used. This would aid comparison of results nationally and ensure any applications for site remediation funding include the same basic information. This could be achieved through the development of a user interface in a software package such as MS Excel that enables the user to input key parameters and obtain severity scores, sub-indices and indices scores, which have been calculated automatically in the background. Such an interface should include the option of exporting data to save it and importing saved data. This could be very similar in functionality to the Highways Agency Water Risk Assessment Tool (HAWRAT) developed by the Highways Agency (now Highways

England) in conjunction with the Environment Agency to assess pollution from road drainage outfalls (UK Government, 2009).

## 5.12 Summary

A novel risk screening assessment method has been proposed that can support coastal landfill managers in identifying which historic coastal landfill sites pose the greatest pollution risk at a national scale for minimal cost by using existing datasets. The highest risk sites can then be prioritised for further investigation, including ground-truthing, or remedial works as appropriate. The risk screening assessment provides a snapshot of the current highest risk sites and should be updated as the underlying datasets are modified to reflect changes to factors such as site condition, e.g. due to maintenance works, flood extent, e.g. due to climate change related sea level rise or changes to defences.

Prior to the assessment being implemented nationally, consultations should be carried out with experts in coastal processes, landfill engineering stability and contamination, and ecology to ensure the parameter severity scores, sub-indices and indices calculations are appropriately weighted to reflect their contribution to the overall risk of historic coastal landfill sites eroding and causing pollution. These also need to be agreed with appropriate regulators, e.g. the Environment Agency and Natural England.

Testing the risk screening assessment, by applying it to eight historic coastal landfills in Essex, found that despite their relatively small sizes, the only two waste filled flood embankments screened (Hadleigh Marsh and Sea Wall in South Fambridge) pose the greatest overall risk of pollution. This is due to their relatively high exposure to drivers of coastal erosion and vulnerability to erosion, which means they are more likely to breach than the other sites screened and, if breached, are likely to release waste at a greater rate than most other sites screened. This means that these two sites should be given priority for expenditure on further investigation and/or remedial actions ahead of the other six sites screened.

Sensitivity testing of the risk screening assessment showed that parameters representing the total landfill volume and contaminant concentrations in the waste have the lowest influence on the overall risk score, and that parameters representing the probability of waste being released, the rate at which it will be released, and the vulnerability of receptors are of much greater importance in determining the overall risk score. This suggests the uncertainty and incompleteness of the data representing the landfill volumes and contaminant concentrations in waste are not of major concern in assessing

the risk of pollution from historic coastal landfill sites, and that resources should not be expended on attempting to improve the accuracy of these parameter datasets, particularly given the difficulties of obtaining representative contaminant data and the high costs involved (see Chapter 3). The importance of the landfill volume and contaminant concentrations in the waste in determining the overall risk score may increase once weightings have been added to the risk screening assessment parameters and indices. However, as most landfilled materials have the potential to be ecologically harmful in some manner regardless of the contaminants they contain, for example by smothering the intertidal zone with their presence in large quantities or harming fauna by being mistaken for food (plastic waste), it is not anticipated that the importance of knowing the contaminant concentrations will increase significantly once weightings have been applied.

# Chapter 6. Conclusions and further research

The overall aim of this research was to investigate the risk of estuarine and/or coastal pollution in the event of historic coastal landfills being inundated or waste being eroded from them. This was achieved by addressing the following three research aims:

1. Determine the potential for contamination of the intertidal zone if solid waste materials are eroded from historic coastal landfills (Chapter 3).
2. Determine the potential for contamination of surface waters by metals released from landfill waste that is either inundated by flood water or eroded into surface waters (Chapter 4).
3. Develop a high-level risk screening assessment methodology that focusses on the risk to the intertidal zone and tidal waters from eroding historic coastal landfills (Chapter 5).

## *6.1 Key findings and outputs*

This research has produced five key findings that advance research into assessing the risk of pollution from historic coastal landfills.

First, this research has found that matrix materials in historic coastal landfills can still contain contaminant concentrations that are comparable to contaminant concentrations in present-day waste, and that are likely to be ecologically harmful if waste erodes and is incorporated into sediments. Concentrations of inorganic contaminants (Cd, Cr, Cu, Pb and Zn) and organic contaminants (acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(ah)anthracene, fluoranthene, fluorene, naphthalene, phenanthrene and pyrene) in waste from the research sites were found to exceed sediment quality guidelines. This means there are likely to be significant adverse biological effects upon bottom feeding and dwelling flora and fauna if waste erodes into intertidal habitats. The ecological risk from eroded waste materials is of particular concern as at least one-third of historic landfills are in or within 100 metres of at least one designated ecological site. The risk of pollution from eroding historic coastal landfills has implications for government expenditure in coastal zone management. It is clear that in the short-term historic coastal landfills and their defences must be maintained to prevent contaminated materials being released and that long-term management strategies need to be developed, which may include removing waste from

the coastal flood zone. Maintaining or improving historic coastal landfill sites solely for environmental protection reasons does not currently meet the criteria for Flood Defence Grant in Aid funding from the government for asset maintenance (J. Lindsay, EA, pers. comm., 12/2/2016), but failure to maintain them could threaten large numbers of designated ecological sites and other sensitive environmental areas. Consideration needs to be given to either amending existing funding criteria to support maintenance and improvement of historic coastal landfills on environmental grounds, or creating new funding mechanisms for this purpose.

Second, this research has found that other solid waste materials, e.g. wood, paper and textiles, can also contain inorganic contaminant concentrations at levels that are likely to be ecologically harmful. This has implications both for landfill risk assessments and for more general contaminated land assessments as industry guidance directs site investigations towards sampling soil (or soil-like materials), water and gases, and does not require assessment of other materials. Guidance for assessing contaminated land, including landfills, should be revised to ensure that the potential presence of other contaminated solid waste materials is also considered.

Third, contaminants in the waste were found to be highly heterogeneous, and significant differences in contaminant concentrations were found between different areas of individual landfills and between landfills. This suggests that existing landfill risk assessments that assume all sites have the same contaminant concentrations are misleading in their conclusions and an alternative approach should be used that better represents differences between sites. The heterogeneity of the contaminants is so great that obtaining representative contaminant datasets is unlikely to be feasible for most sites due to the sampling resolution needed and the associated resource requirements. A more practicable approach to localised studies would be to undertake limited sampling and analyses to determine the types of materials present and obtain indicative contaminant datasets that are likely to be within the same order of magnitude as those from more intensive site investigations and, therefore, would provide a reasonable indication of the pollution potential of a site. When assessing multiple landfills even limited sampling and analyses are likely to be prohibitively expensive. Therefore, for risk screening assessment purposes, this research proposes that risk categories are

assigned based on landfill types to avoid incurring the costs of obtaining contaminant datasets (after Alaska Department of Environmental Conservation, 2015).

Fourth, the proportions of metals released when waste is inundated vary greatly between different subsamples of waste materials in the same leachant, and there are significant differences in the proportions released from waste from different landfill sites. This is probably due to differences in metal concentrations and speciations in the waste, and the presence of insoluble metalliferous particles in variable quantities. This suggests that existing landfill risk assessments that assume all sites release the same proportion of their metal content when inundated are misleading in their conclusions and an alternative approach should be used that better represents differences between sites.

Fifth, significantly higher proportions of metals are released from waste inundated by seawater compared to waste inundated by freshwater (e.g. up to 5,450% for Pb based on median values). This suggests that when climate change effects lead to increased saline intrusion into estuaries there will be increased metal leaching from historic coastal landfill sites that are hydrologically connected to estuarine waters. Although the proportion of the total metal content released from the waste is very low for most metals, the large volumes of waste the sites contain mean that the potential for pollution of surface waters due to the leaching of soluble metals should be considered when assessing risk. If waste does not erode from the landfill during inundation, it is unlikely that surface water quality would be affected by leaching metals, because metals would be attenuated within the waste itself and its capping materials, and diluted in the receiving waters. For the research sites, metals leaching from eroded waste are unlikely to affect surface water quality due to high levels of dilution in the estuary, but for larger sites adjacent to smaller waterbodies there may be adverse effects upon surface water quality if waste erodes. In addition, only a limited suite of metals has been considered and other soluble contaminants, e.g. Ammonium-N, may adversely affect surface water quality. The significant increase in the proportions of metals released in seawater has implications both for assessing the risk of pollution from historic coastal landfills and for legislation and standards relating to testing present-day waste destined for currently operational coastal landfills. Current legislation only requires waste materials to be tested using deionised water leaches, so it is possible that waste is currently being landfilled in vulnerable coastal locations that would not be considered suitable if

legislation were amended to require waste destined for coastal landfills to be tested using artificial seawater leaches.

In addition, the first risk screening assessment that uses existing datasets to rank historic coastal landfill sites by pollution risk has been proposed. No previous ranking method has assessed the risk of pollution from eroding landfills for large numbers of sites without extensive long-term data collection programmes and, until now, there has been no method to determine which of England's 1264 historic coastal landfills pose the greatest environmental risk and, therefore, need prioritizing for maintenance, improvement or future relocation. This is significant as it is likely remediation programmes will need to be phased over many years due to the substantial costs involved. This new assessment method has been developed taking into consideration the key findings detailed above, particularly those relating to the short-comings of existing risk assessments, and considers the risk of pollution from eroding historic coastal landfills based on their exposure to drivers of coastal erosion, their vulnerability to erosion, the waste materials they contain, and the proximity of sensitive environmental and ecological sites. This risk screening assessment has the potential to be used for a national scale risk assessment to determine which historic coastal landfill sites pose the greatest pollution risk and, consequently, should be prioritised for more detailed investigations and remediation as appropriate. Prior to the risk screening assessment being implemented nationally it should be reviewed with appropriate experts and regulators (see recommendations for further research, below).

Sensitivity testing of the risk screening assessment found that parameters representing the probability of waste being released, the rate at which it will be released, and the vulnerability of receptors, are of much greater importance in determining the overall risk score than parameters representing the total landfill volume and contaminant concentrations in the waste. The importance of the landfill volume and contaminant concentrations in the waste in determining the overall risk score may increase once the risk screening assessment method has been reviewed with experts and regulators. However, it is not anticipated that the importance of knowing the contaminant concentrations will increase significantly once weightings have been applied, because most waste has the potential to be ecologically harmful in some manner regardless of the contaminants it contains, e.g. by smothering the intertidal zone with its presence in

large quantities or harming fauna by being mistaken for food (plastic waste). This suggests that resources should not be expended on attempting to improve the accuracy of contaminant datasets, particularly given the difficulties of obtaining representative contaminant data and the high costs involved, and that it is more important when assessing pollution risk to understand the probability of historic coastal landfills eroding and the vulnerability of likely receptors.

## 6.2 Recommendations for further research

A number of ideas for further research have arisen from this work, as outlined below:

1. This research has only assessed the proportion of metals leached at the extremes of the salinity gradient along the catchment to coast continuum. Further research is required to determine how the proportions of metals released vary across the salinity gradient. In addition, further research is required to assess whether the proportions of organic contaminants, e.g. Ammonium-N and PAHs, released during inundation would be as highly variable, and to determine whether they could adversely affect surface water quality.
2. Data relating to landfill volumes are only recorded in local records which are incomplete. To obtain the missing volume data a reliable method, which can be applied consistently across the country, needs to be developed and tested for determining landfill volumes. The addition of landfill volumes to the EA's historic landfill database would support the risk screening assessment and the assessment of the potential for relocating waste away from coastal flood zones.
3. There are currently no data relating to the rate of erosion of solid waste materials from landfill sites or their subsequent dispersion. Knowledge of these would better inform assessment of the likely environmental consequences of solid waste eroding. If the erosion rate is likely to be very low, then it is probable that the risk of pollution from contaminants released into solution or carried on suspended solids is negligible and parameters relating to them could, consequently, be removed from the risk screening assessment. Therefore, a study of the erosion behaviour of waste would better inform the risk assessment process. Understanding the dispersion of waste materials from eroding landfill sites would allow a more accurate assessment of the impact of eroding waste on surrounding environmental and ecological sites, and also support the Environment Agency in the development of appropriate emergency response plans for use in the event of a landfill site being breached.
4. Further research is required to verify whether the order of the ranking applied to the landfill types, which are used as a proxy for contaminant concentrations in the risk assessment, is correct. A study of historic landfill contaminant data from grey literature could potentially be used to investigate this at minimal cost. If

sufficient data exists, it may also be possible to add an additional factor relating to site age into the ranking to account for the variation of materials landfilled in different eras in each landfill type.

5. The risk screening assessment should be reviewed with industry experts before it is used to carry out a national scale assessment of historic coastal landfill sites. Experts on coastal processes, landfill engineering stability and contamination, and ecology should be consulted to determine which parameters, sub-indices and indices, if any, are the most important and should be weighted to ensure the overall risk is sufficiently sensitive to them. The consultations may also identify additional parameters that have not yet been considered or may identify parameters currently included that are superfluous. The method also needs to be agreed with appropriate regulators, e.g. the Environment Agency and Natural England, to ensure they accept the findings of a national scale assessment.
6. The Coastal Vulnerability Index (CVI) for Great Britain that the British Geological Survey is currently developing should be reviewed, when available, to determine whether it provides suitable datasets for use in the risk screening assessment. As use of the CVI will require the purchase of a user-licence, a cost-benefit analysis of its use should be undertaken if it does produce datasets suitable for integration into the historic coastal landfill risk screening assessment.
7. Further research, including cost-benefit analysis, is required into appropriate long-term management options for historic coastal landfills. Currently, removing the waste is prohibitively expensive, but leaving it in situ is unlikely to be a sustainable long-term option.

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# Appendix A. Full datasets for Chapter 3

**Table A.1: Leigh Marshes matrix material moisture content**

| Trial pit 5 | Moisture content (%) | Trial pit 6 | Moisture content (%) |
|---|---|---|---|
| 5A-Digest-A | 78.5 | 6A-Digest-A | 97.2 |
| 5A-Digest-B | 77.5 | 6A-Digest-B | 116.4 |
| 5A-Digest-C | 76.8 | 6A-Digest-C | 107.1 |
| 5B-Digest-A | 75.5 | 6B-Digest-A | 54.8 |
| 5B-Digest-B | 68.3 | 6B-Digest-B | 108.4 |
| 5B-Digest-C | 73.3 | 6B-Digest-C | 83.3 |
| 5C-Digest-A | 66.6 | 6C-Digest-A | 58.6 |
| 5C-Digest-B | 69.6 | 6C-Digest-B | 100.5 |
| 5C-Digest-C | 70.7 | 6C-Digest-C | 90.1 |
| 5D-Digest-A | 71.1 | 6D-Digest-A | 111.2 |
| 5D-Digest-B | 70.1 | 6D-Digest-B | 67.5 |
| 5D-Digest-C | 63.4 | 6D-Digest-C | 77.7 |

**Table A.2: Hadleigh Marsh matrix material moisture content**

| Trial pit 3 | Moisture content (%) | Trial pit 4 | Moisture content (%) |
|---|---|---|---|
| 3A-Digest-A | 45.0 | 4A-Digest-A | 37.0 |
| 3A-Digest-B | 48.7 | 4A-Digest-B | 44.7 |
| 3A-Digest-C | 38.8 | 4A-Digest-C | 193 |
| 3B-Digest-A | 57.5 | 4B-Digest-A | 39.1 |
| 3B-Digest-B | 50.2 | 4B-Digest-B | 37.9 |
| 3B-Digest-C | 55.5 | 4B-Digest-C | 35.9 |
| 3C-Digest-A | 44.1 | 4C-Digest-A | 37.1 |
| 3C-Digest-B | 46.3 | 4C-Digest-B | 39.4 |
| 3C-Digest-C | 41.1 | 4C-Digest-C | 36.9 |
| 3D-Digest-A | 46.4 | 4D-Digest-A | 58.2 |
| 3D-Digest-B | 47.9 | 4D-Digest-B | 43.9 |
| 3D-Digest-C | 45.4 | 4D-Digest-C | 49.3 |

**Table A.3: Hadleigh Marsh wood moisture content**

| Trial pit 3 | Moisture content (%) | Trial pit 4 | Moisture content (%) |
|---|---|---|---|
| 3A-Digest-H | 272.2 | 4D-Digest-K | 101.5 |
| 3A-Digest-J | 202.9 | 4D-Digest-L | 195.7 |
| 3A-Digest-K | 240.2 | 4D-Digest-M | 140.3 |
| 3B-Digest-H | 330.4 | | |
| 3B-Digest-J | 253.1 | | |
| 3B-Digest-K | 243.2 | | |
| 3C-Digest-H | 169 | | |
| 3C-Digest-J | 137.1 | | |
| 3C-Digest-K | 138.7 | | |
| 3D-Digest-H | 196 | | |
| 3D-Digest-J | 196.4 | | |
| 3D-Digest-K | 221.9 | | |

**Table A.4: Hadleigh Marsh paper moisture content**

| Trial pit 3 | Moisture content (%) | Trial pit 4 | Moisture content (%) |
|---|---|---|---|
| 3A-Digest-E | 157.7 | 4A-Digest-D | 260.7 |
| 3A-Digest-F | 137.9 | 4A-Digest-E | 311.5 |
| 3A-Digest-G | 154.4 | 4A-Digest-F | 289.1 |
| 3C-Digest-E | 207.5 | 4C-Digest-D | 193.3 |
| 3C-Digest-F | 194.9 | 4C-Digest-E | 249.2 |
| 3C-Digest-G | 240.7 | 4C-Digest-F | 200.8 |
| 3D-Digest-E | 276.5 | 4D-Digest-D | 300.7 |
| 3D-Digest-F | 214.8 | 4D-Digest-E | 300.7 |
| 3D-Digest-G | 213.4 | 4D-Digest-F | 257.8 |

**Table A.5: Hadleigh Marsh textiles moisture content**

| Trial pit 3 | Moisture content (%) | Trial pit 4 | Moisture content (%) |
|---|---|---|---|
| 3B-Digest-E | 85.3 | 4B-Digest-D | 144.6 |
| 3B-Digest-F | 78.9 | 4B-Digest-E | 214.6 |
| 3B-Digest-G | 64.5 | 4B-Digest-F | 141.8 |
| | | 4D-Digest-G | 101.1 |
| | | 4D-Digest-H | 96.3 |
| | | 4D-Digest-J | 99.3 |

**Table A.6: Leigh Marshes trial pit 5 matrix material inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Limit of detection | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.005 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.60 | 1.11 | 0.05 | 0.17 |
| 5A-Digest-A1 | 5.11 | 25536 | 146 | 985 | 36896 | 4.14 | 43.9 | 106.2 | 899 | 147504 | 1610 | 44.6 | 2065 | 1133 | 2595 | 167 | 2027 | 518 | 1185 |
| 5A-Digest-A2 | 2.15 | 25240 | 172 | 1223 | 27062 | 4.08 | 44.7 | 99.7 | 690 | 161997 | 1771 | 47.4 | 2207 | 1250 | 2599 | 177 | 912 | 532 | 1253 |
| 5A-Digest-A3 | 2.33 | 26412 | 167 | 1455 | 24238 | 4.48 | 46.8 | 100.5 | 972 | 180357 | 1710 | 47.4 | 2957 | 1436 | 2588 | 185 | 2089 | 535 | 1238 |
| 5A-Digest-B1 | 2.32 | 26448 | 105 | 1469 | 24471 | 2.59 | 42.3 | 64.1 | 660 | 101555 | 2033 | 43.6 | 2579 | 878 | 3173 | 149 | 1044 | 528 | 1540 |
| 5A-Digest-B2 | 3.06 | 24556 | 79 | 1312 | 25785 | 3.02 | 42.2 | 61.4 | 1281 | 99797 | 1999 | 43.4 | 2800 | 908 | 3203 | 143 | 831 | 485 | 1628 |
| 5A-Digest-B3 | 2.26 | 26891 | 97 | 1532 | 27542 | 3.19 | 42.3 | 64.0 | 622 | 108449 | 2096 | 44.6 | 2647 | 908 | 3269 | 161 | 1348 | 593 | 2179 |
| 5A-Digest-C1 | 2.18 | 26927 | 107 | 1236 | 23680 | 2.32 | 41.4 | 62.6 | 578 | 113214 | 2410 | 45.7 | 3263 | 1028 | 2819 | 159 | 1381 | 489 | 1566 |
| 5A-Digest-C2 | 1.96 | 29322 | 115 | 1259 | 26311 | 2.87 | 38.5 | 57.9 | 559 | 100012 | 2772 | 49.9 | 3371 | 985 | 2958 | 144 | 638 | 540 | 1357 |
| 5A-Digest-C3 | 1.93 | 26799 | 101 | 1289 | 24188 | 2.92 | 40.5 | 60.7 | 655 | 101801 | 2323 | 46.1 | 2702 | 951 | 2900 | 152 | 625 | 488 | 1439 |
| 5B-Digest-A1 | 1.64 | 29447 | 110 | 1614 | 33483 | 2.43 | 43.7 | 58.9 | 695 | 124058 | 2486 | 47.0 | 2923 | 1080 | 3040 | 157 | 946 | 780 | 1852 |
| 5B-Digest-A2 | 2.03 | 29316 | 134 | 1414 | 24351 | 3.36 | 43.6 | 56.8 | 864 | 132009 | 2498 | 50.8 | 2664 | 1029 | 3228 | 156 | 766 | 1086 | 2663 |
| 5B-Digest-A3 | 1.90 | 24899 | 137 | 1381 | 24131 | 3.93 | 38.5 | 56.9 | 951 | 108224 | 1956 | 42.3 | 2295 | 827 | 2739 | 139 | 678 | 526 | 2524 |
| 5B-Digest-B1 | 1.80 | 29637 | 98 | 1317 | 28999 | 2.40 | 49.3 | 69.5 | 605 | 121386 | 2273 | 46.5 | 3048 | 1044 | 3258 | 170 | 2205 | 649 | 927 |
| 5B-Digest-B2 | 2.03 | 30410 | 109 | 1321 | 29853 | 3.20 | 46.7 | 65.9 | 474 | 119862 | 2631 | 50.1 | 2564 | 1033 | 3207 | 173 | 1059 | 582 | 876 |
| 5B-Digest-B3 | 1.42 | 29347 | 127 | 1529 | 26822 | 3.17 | 44.8 | 57.5 | 353 | 113273 | 2443 | 51.7 | 2434 | 948 | 2957 | 161 | 1810 | 583 | 814 |
| 5B-Digest-C1 | 1.99 | 26986 | 85 | 1232 | 25976 | 2.21 | 42.2 | 61.4 | 438 | 107753 | 2276 | 45.4 | 2243 | 868 | 2889 | 152 | 1908 | 556 | 897 |
| 5B-Digest-C2 | 1.79 | 29507 | 119 | 1384 | 23346 | 2.76 | 42.3 | 64.4 | 447 | 103847 | 2371 | 45.7 | 2337 | 815 | 2939 | 150 | 665 | 601 | 823 |
| 5B-Digest-C3 | 2.30 | 29795 | 109 | 1469 | 29005 | 3.23 | 44.6 | 65.9 | 637 | 121718 | 2427 | 48.3 | 2357 | 895 | 3004 | 160 | 815 | 546 | 942 |
| 5C-Digest-A1 | 4.48 | 30395 | 101 | 1055 | 27816 | 2.36 | 49.6 | 68.2 | 391 | 148147 | 2191 | 49.0 | 2217 | 1083 | 2844 | 183 | 775 | 668 | 897 |
| 5C-Digest-A2 | 4.51 | 24596 | 103 | 1238 | 19772 | 3.13 | 40.8 | 61.9 | 350 | 108909 | 1980 | 40.6 | 1495 | 706 | 2202 | 152 | 1131 | 452 | 963 |
| 5C-Digest-A3 | 2.51 | 21391 | 94 | 1207 | 18616 | 2.88 | 35.3 | 48.7 | 266 | 94164 | 1683 | 36.8 | 1668 | 740 | 1938 | 129 | 1095 | 428 | 1569 |
| 5C-Digest-B1 | 2.45 | 26411 | 101 | 1663 | 23793 | 2.17 | 38.3 | 50.5 | 285 | 99934 | 2256 | 45.8 | 2226 | 882 | 2591 | 132 | 1097 | 613 | 996 |
| 5C-Digest-B2 | 2.57 | 19865 | 81 | 1133 | 19721 | 2.68 | 34.5 | 45.6 | 245 | 102764 | 1647 | 33.2 | 1701 | 762 | 2179 | 125 | 891 | 391 | 906 |
| 5C-Digest-B3 | 2.80 | 28164 | 140 | 1271 | 26543 | 3.03 | 43.8 | 54.6 | 310 | 115430 | 2280 | 44.5 | 2316 | 879 | 2885 | 157 | 1071 | 476 | 998 |
| 5C-Digest-C1 | 2.18 | 33101 | 126 | 1367 | 30811 | 2.67 | 48.0 | 67.8 | 470 | 102654 | 2967 | 53.2 | 2878 | 956 | 3477 | 189 | 1487 | 649 | 1003 |
| 5C-Digest-C2 | 1.56 | 19659 | 67 | 139 | 18797 | 2.53 | 34.4 | 38.9 | 439 | 85267 | 1721 | 35.8 | 1952 | 666 | 2188 | 157 | 2625 | 367 | 667 |
| 5C-Digest-C3 | 2.27 | 30937 | 143 | 1380 | 29170 | 3.57 | 47.7 | 60.1 | 723 | 114865 | 2838 | 51.6 | 2562 | 933 | 3390 | 224 | 839 | 601 | 1097 |
| 5D-Digest-A1 | 3.82 | 30633 | 95 | 1706 | 31117 | 2.10 | 41.1 | 63.3 | 453 | 115438 | 2735 | 54.2 | 3040 | 1013 | 3510 | 163 | 948 | 681 | 1226 |
| 5D-Digest-A2 | 2.26 | 34355 | 143 | 1624 | 29549 | 2.93 | 44.1 | 95.8 | 502 | 114813 | 2924 | 61.7 | 3295 | 1019 | 3756 | 158 | 1312 | 640 | 1336 |
| 5D-Digest-A3 | 2.03 | 31242 | 166 | 321 | 24706 | 3.03 | 43.0 | 69.0 | 560 | 114605 | 2656 | 61.6 | 3099 | 957 | 3698 | 161 | 763 | 650 | 1268 |
| 5D-Digest-B1 | 2.72 | 33924 | 113 | 1827 | 27831 | 3.15 | 41.0 | 63.4 | 563 | 110640 | 2925 | 58.7 | 3115 | 885 | 3508 | 146 | 1727 | 510 | 1675 |
| 5D-Digest-B2 | 2.85 | 30197 | 126 | 327 | 25687 | 3.06 | 41.9 | 61.2 | 1030 | 95845 | 2551 | 57.9 | 2914 | 869 | 3566 | 161 | 1552 | 536 | 1201 |
| 5D-Digest-B3 | 1.87 | 29957 | 147 | 1421 | 27687 | 3.23 | 40.3 | 58.6 | 436 | 97330 | 2656 | 52.1 | 3099 | 851 | 3361 | 151 | 1038 | 576 | 1080 |
| 5D-Digest-C1 | 2.13 | 30381 | 114 | 1586 | 30227 | 2.68 | 41.0 | 63.2 | 621 | 105227 | 2761 | 53.3 | 3268 | 871 | 3226 | 151 | 1137 | 536 | 2115 |
| 5D-Digest-C2 | 2.23 | 27851 | 141 | 275 | 29944 | 3.94 | 39.4 | 59.1 | 884 | 90137 | 2617 | 52.4 | 3374 | 1064 | 3416 | 150 | 1513 | 534 | 3100 |
| 5D-Digest-C3 | 1.70 | 26167 | 105 | 1316 | 24357 | 3.19 | 36.8 | 60.9 | 928 | 76920 | 2410 | 47.1 | 3120 | 810 | 2973 | 155 | 1210 | 447 | 1758 |

**Table A.7: Leigh Marshes trial pit 6 matrix material inorganic contaminant concentrations**

| All data mg kg-1 | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Limit of detection | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.72 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.60 | 1.11 | 0.05 | 0.17 |
| 6A-Digest-A1 | 4.05 | 17450 | 59 | 729 | 19912 | 7.79 | 27.9 | 72.5 | 559 | 178208 | 1310 | 28.3 | 1742 | 842 | 871 | 138.6 | 1332 | 317 | 1691 |
| 6A-Digest-A2 | 2.46 | 17209 | 125 | 1404 | 23050 | 8.23 | 27.7 | 76.7 | 346 | 205288 | 1500 | 29.4 | 1928 | 839 | 877 | 140.6 | 1070 | 268 | 1445 |
| 6A-Digest-A3 | 8.48 | 15854 | 154 | 1129 | 19681 | 8.21 | 26.8 | 92.5 | 951 | 210530 | 1451 | 29.1 | 1770 | 851 | 989 | 135.2 | 1421 | 261 | 1508 |
| 6A-Digest-B1 | 3.32 | 16570 | 83 | 1323 | 22769 | 10.07 | 21.0 | 108.7 | 647 | 165477 | 1362 | 23.6 | 1910 | 717 | 644 | 100.1 | 2986 | 277 | 2557 |
| 6A-Digest-B2 | 10.39 | 17999 | 72 | 531 | 21266 | 8.64 | 21.1 | 112.3 | 818 | 185464 | 1258 | 23.4 | 1751 | 804 | 738 | 111.5 | 3470 | 320 | 2546 |
| 6A-Digest-B3 | 1.58 | 16043 | 68 | 1199 | 23702 | 8.81 | 24.0 | 91.8 | 856 | 196627 | 1091 | 21.3 | 1574 | 825 | 685 | 119.6 | 3254 | 279 | 1934 |
| 6A-Digest-C1 | 5.79 | 16867 | 62 | 1497 | 13303 | 5.28 | 15.1 | 94.0 | 665 | 94606 | 1878 | 24.5 | 2206 | 433 | 420 | 99.5 | 1280 | 180 | 1623 |
| 6A-Digest-C2 | 1.91 | 18166 | 136 | 1358 | 11507 | 4.89 | 16.0 | 86.2 | 292 | 89773 | 2035 | 26.5 | 2236 | 435 | 526 | 78.4 | 1067 | 161 | 1623 |
| 6A-Digest-C3 | 1.72 | 17031 | 39 | 1207 | 13688 | 5.49 | 15.3 | 77.8 | 258 | 87293 | 1974 | 26.2 | 2282 | 468 | 462 | 78.5 | 1248 | 163 | 1866 |
| 6B-Digest-A1 | 2.51 | 27350 | 66 | 1196 | 36155 | 4.17 | 38.1 | 53.0 | 331 | 109612 | 2333 | 46.2 | 3453 | 1083 | 2747 | 143.2 | 14936 | 582 | 1986 |
| 6B-Digest-A2 | 2.51 | 25775 | 44 | 1101 | 34126 | 4.40 | 36.1 | 54.0 | 443 | 111073 | 2177 | 46.3 | 2873 | 938 | 2580 | 136.4 | 1320 | 472 | 936 |
| 6B-Digest-A3 | 1.48 | 12815 | <LOD | 141 | 27882 | 1.64 | 38.1 | 12.6 | 803 | 42710 | 1692 | 37.4 | 2917 | 870 | 2294 | 112.3 | 1362 | 387 | 977 |
| 6B-Digest-B1 | 2.97 | 15312 | 7 | 903 | 19082 | 9.95 | 29.1 | 159.4 | 1535 | 153918 | 1475 | 26.2 | 1852 | 807 | 918 | 125.1 | 6825 | 359 | 3927 |
| 6B-Digest-B2 | 6.30 | 15575 | <LOD | 1433 | 22893 | 9.31 | 23.6 | 152.8 | 732 | 136870 | 1571 | 26.9 | 1971 | 857 | 1075 | 107.4 | 6957 | 353 | 4392 |
| 6B-Digest-B3 | 3.42 | 14120 | 13 | 669 | 21824 | 6.77 | 26.6 | 135.4 | 961 | 154710 | 1364 | 27.1 | 1855 | 876 | 939 | 125.4 | 8380 | 375 | 4549 |
| 6B-Digest-C1 | 3.68 | 20928 | 8 | 363 | 30846 | 7.28 | 26.7 | 95.7 | 332 | 154256 | 1922 | 35.3 | 2599 | 840 | 1321 | 127.2 | 8321 | 369 | 4331 |
| 6B-Digest-C2 | 253.64 | 21234 | 58 | 414 | 25970 | 8.26 | 28.0 | 96.4 | 295 | 170084 | 1831 | 35.9 | 2492 | 900 | 1484 | 125.1 | 5293 | 343 | 5420 |
| 6B-Digest-C3 | 3.41 | 20147 | 18 | 658 | 27975 | 6.48 | 27.3 | 98.1 | 262 | 130793 | 1867 | 36.2 | 2408 | 837 | 1384 | 114.8 | 8891 | 395 | 5731 |
| 6C-Digest-A1 | 2.05 | 30575 | 65 | 1070 | 38040 | 5.71 | 37.7 | 59.0 | 634 | 114819 | 2606 | 53.0 | 3578 | 1186 | 3064 | 139.4 | 886 | 500 | 1105 |
| 6C-Digest-A2 | 1.62 | 31833 | 55 | 1241 | 39280 | 7.07 | 38.8 | 59.6 | 309 | 124709 | 2351 | 54.2 | 3054 | 1132 | 2847 | 133.9 | 1014 | 526 | 1017 |
| 6C-Digest-A3 | 1.55 | 32121 | 62 | 1168 | 41292 | 6.01 | 35.0 | 54.8 | 303 | 112929 | 2372 | 54.7 | 3594 | 1085 | 3188 | 127.8 | 1110 | 520 | 1262 |
| 6C-Digest-B1 | 5.14 | 20026 | 74 | 402 | 30514 | 6.60 | 29.6 | 98.4 | 20140 | 114194 | 2338 | 36.6 | 2757 | 859 | 1695 | 119.4 | 2439 | 341 | 1922 |
| 6C-Digest-B1 | No results – sample spilt | | | | | | | | | | | | | | | | | | |
| 6C-Digest-B3 | 5.19 | 19111 | 58 | 478 | 29500 | 7.62 | 24.8 | 96.3 | 15766 | 99228 | 1927 | 33.8 | 2787 | 983 | 1329 | 101.2 | 4130 | 336 | 1936 |
| 6C-Digest-C1 | 2.14 | 17635 | 52 | 84 | 21998 | 10.95 | 24.3 | 63.2 | 2671 | 153559 | 1603 | 32.3 | 2067 | 740 | 954 | 113.0 | 1293 | 328 | 1242 |
| 6C-Digest-C2 | 3.09 | 18435 | 32 | 789 | 20704 | 6.36 | 20.6 | 98.5 | 4660 | 121631 | 1800 | 31.0 | 1920 | 597 | 896 | 102.8 | 4790 | 382 | 1505 |
| 6C-Digest-C3 | 2.38 | 21238 | 52 | 1480 | 22264 | 7.78 | 27.5 | 83.9 | 2419 | 162333 | 1887 | 34.6 | 2206 | 864 | 1329 | 140.6 | 2031 | 442 | 1523 |
| 6D-Digest-A1 | 2.67 | 20356 | 8 | 879 | 22967 | 6.80 | 28.2 | 61.1 | 519 | 127962 | 2186 | 33.8 | 2287 | 768 | 1389 | 124.3 | 1552 | 309 | 1722 |
| 6D-Digest-A2 | 2.40 | 19772 | 48 | 1249 | 24530 | 7.33 | 26.7 | 57.2 | 429 | 124307 | 2122 | 34.5 | 2230 | 741 | 1388 | 118.8 | 1170 | 359 | 1509 |
| 6D-Digest-A3 | 2.15 | 20386 | 22 | 1378 | 21779 | 6.39 | 30.0 | 62.5 | 423 | 134732 | 2157 | 36.7 | 2152 | 745 | 1428 | 129.4 | 1261 | 303 | 1697 |
| 6D-Digest-B1 | 3.15 | 23197 | 28 | 1023 | 22749 | 8.25 | 33.2 | 65.6 | 830 | 85803 | 2446 | 41.3 | 3069 | 852 | 1633 | 134.3 | 1898 | 325 | 2980 |
| 6D-Digest-B2 | 2.45 | 25284 | 52 | 652 | 23568 | 9.19 | 33.2 | 75.0 | 412 | 80190 | 2736 | 43.9 | 3278 | 920 | 1655 | 134.8 | 4313 | 367 | 3520 |
| 6D-Digest-B3 | 4.12 | 26791 | 38 | 245 | 24553 | 6.46 | 33.1 | 70.6 | 409 | 88298 | 2807 | 46.9 | 3343 | 813 | 1782 | 132.8 | 1438 | 343 | 3151 |
| 6D-Digest-C1 | 6.07 | 18803 | <LOD | 1086 | 17372 | 5.62 | 25.1 | 71.8 | 3227 | 149634 | 2098 | 31.1 | 2857 | 746 | 718 | 139.2 | 2167 | 235 | 1694 |
| 6D-Digest-C2 | 16.29 | 20698 | 30 | 611 | 24734 | 6.55 | 30.5 | 86.3 | 778 | 139711 | 2241 | 35.6 | 2746 | 778 | 1200 | 148.7 | 2607 | 299 | 2015 |
| 6D-Digest-C3 | 5.55 | 19419 | 27 | 683 | 19250 | 6.21 | 30.1 | 105.8 | 2081 | 145902 | 2082 | 33.3 | 2565 | 838 | 885 | 162.0 | 2501 | 265 | 1966 |

**Table A.8: Hadleigh Marsh trial pit 3 matrix material inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Limit of detection* | *0.06* | *0.12* | *5.83* | *0.09* | *0.16* | *0.005* | *0.31* | *0.12* | *0.08* | *6.55* | *0.17* | *0.03* | *0.09* | *0.07* | *0.07* | *0.60* | *1.11* | *0.05* | *0.17* |
| 3A-Digest-A1 | <LOD | 46366 | 110 | 110.2 | 8289 | 0.81 | 35.1 | 71.8 | 25.1 | 63117 | 8788 | 80.4 | 11998 | 2510 | 797 | 65.0 | 45 | 85 | 185 |
| 3A-Digest-A2 | <LOD | 43018 | 100 | 100.0 | 7413 | 1.34 | 24.9 | 78.6 | 28.9 | 79049 | 7975 | 79.1 | 11085 | 1124 | 842 | 70.4 | 54 | 83 | 462 |
| 3A-Digest-A3 | <LOD | 39975 | 91 | 90.7 | 8605 | 1.05 | 20.1 | 69.1 | 21.2 | 57620 | 7759 | 73.4 | 11627 | 1089 | 751 | 55.7 | 62 | 79 | 247 |
| 3A-Digest-B1 | 0.30 | 21429 | 213 | 212.8 | 18568 | 20.58 | <LOD | 63.3 | <LOD | 92564 | 3084 | <LOD | 4345 | 507 | 765 | <LOD | 273 | 109 | 1109 |
| 3A-Digest-B2 | <LOD | 22782 | 202 | 201.6 | 20960 | 9.69 | 11.7 | 43.5 | 85.8 | 131132 | 3596 | 32.5 | 4983 | 498 | 875 | 74.7 | 252 | 110 | 1064 |
| 3A-Digest-B3 | <LOD | 22196 | 461 | 461.2 | 20788 | 4.37 | 10.5 | 50.2 | 102.6 | 105188 | 3038 | 29.0 | 3843 | 422 | 1094 | 61.5 | 338 | 125 | 1845 |
| 3A-Digest-C1 | <LOD | 30732 | 93 | 92.9 | 20268 | 0.87 | 10.3 | 50.7 | 23.5 | 42579 | 5785 | 55.5 | 8813 | 290 | 714 | 35.0 | 91 | 100 | 293 |
| 3A-Digest-C2 | <LOD | 29995 | 71 | 70.6 | 21542 | 0.75 | 11.3 | 51.6 | 22.8 | 39113 | 5665 | 59.8 | 9326 | 319 | 830 | 35.5 | 39 | 99 | 199 |
| 3A-Digest-C3 | <LOD | 33399 | 78 | 78.2 | 24449 | 0.80 | 10.9 | 53.7 | 22.6 | 40451 | 6299 | 60.6 | 9122 | 304 | 825 | 34.9 | 48 | 115 | 209 |
| 3B-Digest-A1 | 0.72 | 37624 | 117 | 117.0 | 14745 | 0.61 | 15.8 | 68.7 | 29.6 | 45469 | 7298 | 68.4 | 10231 | 414 | 910 | 45.3 | 104 | 91 | 314 |
| 3B-Digest-A2 | <LOD | 16682 | 293 | 292.8 | 19807 | 4.42 | 20.1 | 68.3 | 79.6 | 50068 | 3194 | 37.8 | 5167 | 608 | 824 | 61.5 | 508 | 205 | 3272 |
| 3B-Digest-A3 | <LOD | 28280 | 332 | 332.5 | 14852 | 1.93 | 19.6 | 80.6 | 61.1 | 66621 | 5065 | 49.9 | 6602 | 476 | 1113 | 60.9 | 494 | 112 | 926 |
| 3B-Digest-B1 | <LOD | 20101 | 249 | 248.9 | 32920 | 0.88 | 12.9 | 26.1 | 37.2 | 22971 | 2217 | 65.3 | 7239 | 716 | 891 | 31.7 | 300 | 247 | 672 |
| 3B-Digest-B2 | 6.00 | 27076 | 274 | 274.2 | 28300 | 2.71 | 16.6 | 64.6 | 86.0 | 81221 | 4193 | 41.2 | 4965 | 738 | 869 | 98.5 | 460 | 139 | 5128 |
| 3B-Digest-B3 | <LOD | 23898 | 227 | 226.7 | 15616 | 2.17 | 20.3 | 58.9 | 88.2 | 95973 | 4408 | 45.6 | 6358 | 550 | 697 | 67.7 | 383 | 114 | 1429 |
| 3B-Digest-C1 | No results – sample spilt | | | | | | | | | | | | | | | | | | |
| 3B-Digest-C2 | <LOD | 25258 | 299 | 299.3 | 15064 | 1.64 | 16.1 | 78.5 | 60.5 | 83580 | 4253 | 41.8 | 5527 | 554 | 873 | 59.1 | 317 | 105 | 1277 |
| 3B-Digest-C3 | <LOD | 15440 | 188 | 188.5 | 11983 | 2.06 | 12.5 | 50.4 | 52.2 | 81769 | 2911 | 28.6 | 3950 | 466 | 564 | 50.7 | 297 | 79 | 789 |
| 3C-Digest-A1 | 3.74 | 44388 | 79 | 78.9 | 9241 | 0.44 | 18.4 | 72.0 | 23.0 | 53810 | 8175 | 82.6 | 11549 | 643 | 780 | 55.1 | 27 | 82 | 116 |
| 3C-Digest-A2 | <LOD | 33366 | 76 | 76.3 | 7239 | 0.88 | 23.6 | 64.3 | 21.4 | 54382 | 6097 | 67.3 | 10489 | 1040 | 656 | 57.3 | 47 | 70 | 136 |
| 3C-Digest-A3 | <LOD | 43278 | 96 | 96.2 | 7683 | 0.91 | 23.1 | 74.2 | 25.2 | 55504 | 8182 | 83.7 | 11532 | 916 | 802 | 50.6 | 55 | 85 | 138 |
| 3C-Digest-B1 | <LOD | 20963 | 60 | 59.9 | 6714 | 0.35 | 15.3 | 40.7 | 16.7 | 27346 | 3999 | 48.7 | 8016 | 471 | 527 | 35.4 | 44 | 58 | 137 |
| 3C-Digest-B2 | <LOD | 40197 | 99 | 98.9 | 8104 | 0.98 | 18.9 | 70.3 | 56.6 | 55652 | 7374 | 80.3 | 10953 | 893 | 827 | 50.7 | 66 | 80 | 241 |
| 3C-Digest-B3 | <LOD | 36214 | 93 | 93.0 | 8805 | 1.01 | 18.6 | 67.1 | 23.3 | 52601 | 6613 | 71.5 | 10163 | 708 | 671 | 46.8 | 73 | 80 | 220 |
| 3C-Digest-C1 | <LOD | 38163 | 81 | 80.5 | 15217 | 0.46 | 17.3 | 67.4 | 16.7 | 43657 | 7218 | 77.7 | 10434 | 631 | 862 | 45.2 | 30 | 100 | 99 |
| 3C-Digest-C2 | <LOD | 38829 | 74 | 73.8 | 16410 | 0.74 | 16.0 | 63.4 | 16.8 | 42580 | 7336 | 73.0 | 10657 | 488 | 749 | 43.9 | 31 | 95 | 128 |
| 3C-Digest-C3 | <LOD | 42713 | 81 | 80.7 | 16962 | 0.74 | 15.4 | 65.9 | 16.8 | 44950 | 8381 | 76.2 | 11467 | 408 | 798 | 42.7 | 29 | 98 | 140 |
| 3D-Digest-A1 | <LOD | 33454 | 130 | 130.0 | 11024 | 1.39 | 15.3 | 59.7 | 33.3 | 40001 | 6430 | 61.8 | 9735 | 405 | 883 | 45.9 | 159 | 86 | 447 |
| 3D-Digest-A2 | <LOD | 34862 | 148 | 148.4 | 11127 | 1.22 | 16.3 | 67.0 | 32.4 | 50013 | 6536 | 60.9 | 9238 | 406 | 880 | 48.6 | 270 | 84 | 358 |
| 3D-Digest-A3 | <LOD | 13736 | 119 | 119.0 | 6920 | 0.78 | 9.4 | 34.0 | 20.7 | 23912 | 2893 | 30.2 | 5320 | 219 | 482 | 30.4 | 100 | 48 | 221 |
| 3D-Digest-B1 | <LOD | 36991 | 92 | 91.8 | 11415 | 0.52 | 16.7 | 62.5 | 24.2 | 46253 | 7247 | 67.1 | 10392 | 331 | 861 | 43.6 | 81 | 79 | 256 |
| 3D-Digest-B2 | <LOD | 35802 | 130 | 130.5 | 14664 | 1.42 | 21.5 | 59.1 | 34.7 | 58364 | 6843 | 64.2 | 9750 | 504 | 883 | 50.0 | 146 | 101 | 387 |
| 3D-Digest-B3 | <LOD | 34131 | 129 | 128.7 | 11113 | 1.18 | 18.8 | 59.8 | 39.7 | 45112 | 6634 | 61.4 | 9355 | 367 | 884 | 46.5 | 117 | 83 | 418 |
| 3D-Digest-C1 | <LOD | 33656 | 114 | 113.7 | 12145 | 0.73 | 15.2 | 60.0 | 25.1 | 43267 | 6520 | 61.4 | 10207 | 321 | 848 | 45.2 | 108 | 82 | 434 |
| 3D-Digest-C2 | <LOD | 32663 | 106 | 105.9 | 10797 | 1.00 | 16.8 | 59.2 | 26.9 | 46537 | 6692 | 57.7 | 10990 | 352 | 784 | 46.5 | 104 | 74 | 348 |
| 3D-Digest-C3 | <LOD | 38683 | 87 | 87.2 | 9822 | 1.01 | 15.4 | 62.7 | 22.2 | 47443 | 7449 | 67.5 | 11306 | 317 | 864 | 44.4 | 68 | 77 | 302 |

**Table A.9: Hadleigh Marsh trial pit 4 matrix material inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Limit of detection | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.005 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.60 | 1.11 | 0.05 | 0.17 |
| 4A-Digest-A1 | <LOD | 17692 | <LOD | 100 | 14625 | 3.73 | 8.7 | 43.5 | 53.8 | 40126 | 2301 | 19.1 | 2766 | 248 | 366 | 30.7 | 117.1 | 72.1 | 154 |
| 4A-Digest-A2 | <LOD | 20362 | 10.1 | 139 | 12963 | 16.60 | 9.2 | 40.6 | 71.5 | 49905 | 2399 | 20.0 | 2942 | 340 | 428 | 33.5 | 106.3 | 99.5 | 195 |
| 4A-Digest-A3 | <LOD | 17935 | 29.7 | 99 | 12824 | 7.48 | 7.7 | 40.2 | 84.8 | 48648 | 2516 | 22.9 | 3231 | 308 | 444 | 34.2 | 103.0 | 73.6 | 326 |
| 4A-Digest-B1 | <LOD | 29098 | 30.2 | 80 | 6764 | 2.63 | 15.5 | 49.9 | 31.3 | 40418 | 4732 | 40.5 | 7145 | 708 | 456 | 33.7 | 35.3 | 55.5 | 93 |
| 4A-Digest-B2 | 233.24 | 22625 | <LOD | 64 | 5207 | 2.30 | 8.1 | 38.6 | 24.3 | 26703 | 2515 | 22.3 | 3321 | 206 | 288 | 22.1 | 61.6 | 40.5 | 62 |
| 4A-Digest-B3 | 0.56 | 20123 | 6.6 | 92 | 6040 | 4.35 | 10.0 | 39.5 | 45.5 | 27646 | 2539 | 23.2 | 3061 | 304 | 310 | 32.7 | 68.4 | 42.9 | 114 |
| 4A-Digest-C1 | 1.52 | 11575 | 7.0 | 305 | 14024 | 15.96 | 12.0 | 65.8 | 201.0 | 85308 | 1171 | 15.7 | 2166 | 607 | 665 | 127.8 | 199.4 | 106.5 | 1316 |
| 4A-Digest-C2 | 0.47 | 16223 | 12.6 | 445 | 9578 | 10.97 | 7.1 | 58.4 | 114.2 | 39962 | 1496 | 21.0 | 2791 | 277 | 578 | 42.9 | 154.0 | 81.3 | 962 |
| 4A-Digest-C3 | 2.92 | 13568 | <LOD | 611 | 10188 | 16.41 | 6.3 | 49.3 | 157.9 | 54515 | 1178 | 14.0 | 1886 | 310 | 550 | 42.2 | 136.3 | 114.0 | 2532 |
| 4B-Digest-A1 | <LOD | 36301 | 31.5 | 80 | 13068 | 0.64 | 17.7 | 71.1 | 31.9 | 52348 | 6685 | 66.0 | 12602 | 579 | 358 | 54.8 | 45.6 | 81.1 | 70 |
| 4B-Digest-A2 | <LOD | 46378 | 52.6 | 95 | 17804 | 0.96 | 17.4 | 73.4 | 28.6 | 51541 | 8262 | 74.5 | 13345 | 574 | 435 | 52.1 | 36.0 | 100.1 | 302 |
| 4B-Digest-A3 | 1.24 | 42940 | 31.3 | 89 | 10857 | 1.03 | 17.0 | 69.1 | 60.4 | 63197 | 7267 | 67.8 | 11770 | 577 | 354 | 52.1 | 39.0 | 72.7 | 64 |
| 4B-Digest-B1 | <LOD | 30496 | 22.0 | 667 | 9654 | 0.67 | 17.8 | 58.6 | 35.6 | 48692 | 5351 | 51.0 | 9686 | 381 | 283 | 53.4 | 48.1 | 107.3 | 86 |
| 4B-Digest-B2 | <LOD | 20469 | <LOD | 746 | 8451 | 6.51 | 19.1 | 44.9 | 28.8 | 45949 | 3506 | 33.3 | 5908 | 283 | 193 | 56.0 | 55.2 | 94.6 | 213 |
| 4B-Digest-B3 | <LOD | 27677 | 18.8 | 1516 | 7981 | 0.79 | 17.3 | 50.9 | 30.7 | 40544 | 4489 | 40.2 | 6471 | 275 | 257 | 52.3 | 63.0 | 167.3 | 83 |
| 4B-Digest-C1 | <LOD | 39152 | 39.7 | 93 | 11116 | 0.54 | 13.7 | 64.8 | 38.0 | 48571 | 6190 | 57.6 | 9793 | 422 | 353 | 47.0 | 49.9 | 70.7 | 108 |
| 4B-Digest-C2 | <LOD | 23295 | 9.3 | 74 | 5764 | 0.47 | 11.7 | 41.8 | 38.7 | 23312 | 3880 | 39.1 | 6235 | 246 | 277 | 39.0 | 48.8 | 48.2 | 61 |
| 4B-Digest-C3 | <LOD | 63428 | 9.1 | 76 | 19270 | 0.79 | 11.3 | 46.2 | 103.0 | 100225 | 10971 | 34.2 | 16567 | 273 | 251 | 44.4 | 59.4 | 65.2 | 199 |
| 4C-Digest-A1 | <LOD | 24876 | <LOD | 114 | 5019 | 1.08 | 12.9 | 43.0 | 32.0 | 51032 | 5009 | 35.5 | 6263 | 521 | 271 | 39.1 | 38.9 | 53.2 | 111 |
| 4C-Digest-A2 | <LOD | 26700 | <LOD | 72 | 5037 | 1.29 | 12.3 | 43.1 | 32.1 | 48043 | 5221 | 37.5 | 6196 | 496 | 311 | 36.1 | 38.3 | 52.9 | 65 |
| 4C-Digest-A3 | <LOD | 28639 | 8.2 | 72 | 5787 | 2.10 | 13.2 | 43.9 | 36.1 | 49031 | 5499 | 39.4 | 6287 | 504 | 338 | 38.0 | 41.3 | 56.0 | 108 |
| 4C-Digest-B1 | <LOD | 28494 | <LOD | 91 | 5992 | 1.16 | 13.2 | 45.0 | 28.4 | 48270 | 5458 | 39.2 | 6440 | 690 | 307 | 35.8 | 43.7 | 59.8 | 87 |
| 4C-Digest-B2 | <LOD | 29829 | <LOD | 156 | 4813 | 1.61 | 13.0 | 47.5 | 30.3 | 47964 | 5505 | 40.3 | 6365 | 580 | 274 | 36.0 | 59.1 | 56.7 | 199 |
| 4C-Digest-B3 | <LOD | 32809 | <LOD | 103 | 5084 | 2.98 | 12.7 | 55.5 | 39.0 | 49310 | 5990 | 45.5 | 6570 | 569 | 326 | 43.5 | 57.5 | 59.9 | 205 |
| 4C-Digest-C1 | <LOD | 36740 | <LOD | 336 | 5422 | 0.35 | 14.1 | 64.7 | 24.1 | 39107 | 6600 | 58.7 | 7924 | 202 | 238 | 40.7 | 21.8 | 58.4 | 35 |
| 4C-Digest-C2 | <LOD | 32422 | <LOD | 275 | 7216 | 0.82 | 15.1 | 70.6 | 43.1 | 44669 | 5713 | 47.2 | 7187 | 266 | 237 | 50.0 | 30.3 | 55.1 | 181 |
| 4C-Digest-C3 | <LOD | 36128 | <LOD | 159 | 6288 | 0.84 | 13.9 | 63.8 | 29.0 | 42429 | 6563 | 52.8 | 7925 | 241 | 244 | 46.6 | 28.0 | 56.5 | 56 |
| 4D-Digest-A1 | <LOD | 31854 | <LOD | 112 | 8723 | 27.50 | 13.1 | 53.3 | 57.6 | 40903 | 6352 | 51.0 | 7515 | 241 | 469 | 41.4 | 65.5 | 63.2 | 234 |
| 4D-Digest-A2 | <LOD | 30428 | 15.7 | 130 | 12276 | 25.76 | 15.7 | 370.5 | 132.0 | 45820 | 5736 | 45.5 | 6991 | 310 | 523 | 355.8 | 88.4 | 87.3 | 303 |
| 4D-Digest-A3 | <LOD | 31164 | <LOD | 72 | 9042 | 5.71 | 12.6 | 51.4 | 322.3 | 37961 | 6586 | 50.0 | 8724 | 206 | 426 | 37.4 | 31.3 | 57.5 | 119 |
| 4D-Digest-B1 | <LOD | 27411 | <LOD | 78 | 22998 | 2.00 | 18.0 | 55.7 | 47.2 | 82832 | 5487 | 40.0 | 6241 | 651 | 445 | 147.5 | 43.9 | 110.8 | 133 |
| 4D-Digest-B2 | <LOD | 28560 | 35.0 | 84 | 13483 | 1.79 | 16.5 | 119.4 | 76.0 | 71187 | 5668 | 41.6 | 6302 | 709 | 456 | 209.1 | 87.0 | 79.7 | 175 |
| 4D-Digest-B3 | <LOD | 28801 | <LOD | 96 | 13223 | 1.86 | 14.6 | 62.7 | 42.9 | 66806 | 5689 | 41.9 | 6413 | 700 | 382 | 139.8 | 49.6 | 78.0 | 128 |
| 4D-Digest-C1 | <LOD | 27028 | <LOD | 94 | 12117 | 8.71 | 11.6 | 52.6 | 30.1 | 43079 | 5713 | 42.2 | 8048 | 225 | 368 | 37.9 | 56.9 | 66.0 | 166 |
| 4D-Digest-C2 | <LOD | 34519 | <LOD | 118 | 13119 | 1.52 | 12.5 | 55.9 | 44.9 | 48533 | 6995 | 54.0 | 8919 | 278 | 559 | 41.4 | 76.8 | 80.8 | 260 |
| 4D-Digest-C3 | <LOD | 27078 | <LOD | 125 | 12635 | 1.14 | 11.5 | 46.0 | 39.4 | 35548 | 5541 | 46.5 | 8033 | 267 | 435 | 43.8 | 81.7 | 75.7 | 230 |

**Table A.10: Hadleigh Marsh trial pit 3 wood inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Limit of detection** | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.72 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.60 | 1.11 | 0.05 | 0.17 |
| **3A-Digest-H1** | <LOD | 374 | <LOD | 11.0 | 2300 | <LOD | <LOD | 17.3 | 17.0 | 30598 | 403 | <LOD | 703 | 37.5 | 680 | 6.9 | 25 | 9.9 | 174 |
| **3A-Digest-H2** | <LOD | 296 | <LOD | 20.4 | 2717 | <LOD | <LOD | 15.7 | 18.8 | 11186 | 378 | <LOD | 750 | 28.6 | 803 | <LOD | 30 | 12.8 | 153 |
| **3A-Digest-H3** | <LOD | 1893 | <LOD | 109.3 | 3880 | <LOD | <LOD | 24.9 | 23.5 | 35886 | 821 | <LOD | 1084 | 61.1 | 1004 | 9.8 | 52 | 19.3 | 353 |
| **3A-Digest-J1** | 1.05 | 699 | <LOD | 28.0 | 1977 | <LOD | <LOD | 52.8 | 39.2 | 195573 | 418 | <LOD | 524 | 222.1 | 561 | <LOD | 51 | 12.1 | 561 |
| **3A-Digest-J2** | <LOD | 2201 | <LOD | 42.1 | 2519 | <LOD | <LOD | 48.6 | 59.3 | 317972 | 517 | <LOD | 834 | 341.3 | 601 | <LOD | <LOD | 15.0 | 759 |
| **3A-Digest-J3** | <LOD | 1444 | <LOD | 29.2 | 5342 | <LOD | <LOD | 5.8 | 60.0 | 12396 | 824 | <LOD | 1189 | 72.4 | 743 | 8.1 | 937 | 19.2 | 953 |
| **3A-Digest-K1** | <LOD | 3579 | <LOD | 75.1 | 6211 | <LOD | <LOD | 12.4 | 48.9 | 52558 | 1221 | <LOD | 1473 | 122.9 | 829 | 19.1 | 850 | 26.2 | 2380 |
| **3A-Digest-K2** | <LOD | 5799 | <LOD | 136.9 | 5785 | <LOD | 7.0 | 24.1 | 95.8 | 37927 | 1915 | <LOD | 1821 | 167.5 | 758 | 24.3 | 746 | 30.7 | 1087 |
| **3A-Digest-K3** | <LOD | 3459 | <LOD | 75.3 | 7284 | <LOD | <LOD | 13.0 | 65.2 | 31297 | 1222 | <LOD | 1615 | 116.6 | 941 | 17.6 | 1180 | 30.0 | 1277 |
| **3B-Digest-H1** | <LOD | 2143 | 23.8 | 144.1 | 13217 | <LOD | 5.3 | 11.3 | 84.1 | 30040 | 1256 | <LOD | 1992 | 162.2 | 999 | 14.3 | 484 | 77.1 | 1509 |
| **3B-Digest-H2** | <LOD | 1786 | <LOD | 137.8 | 12558 | <LOD | 5.8 | 9.4 | 85.3 | 20853 | 1138 | <LOD | 1906 | 112.2 | 814 | 20.9 | 609 | 68.3 | 1250 |
| **3B-Digest-H3** | <LOD | 1513 | <LOD | 86.5 | 13539 | <LOD | 6.6 | 9.1 | 83.3 | 13045 | 1164 | <LOD | 2048 | 156.3 | 987 | 13.2 | 198 | 65.2 | 1094 |
| **3B-Digest-J1** | <LOD | 4169 | <LOD | 136.6 | 9140 | <LOD | 7.6 | 18.1 | 36.4 | 19086 | 1533 | <LOD | 1857 | 200.4 | 697 | 23.2 | 293 | 39.7 | 1279 |
| **3B-Digest-J2** | <LOD | 1987 | 9.9 | 89.7 | 9969 | <LOD | 8.0 | 10.4 | 43.2 | 7128 | 1171 | <LOD | 1578 | 166.5 | 783 | 13.1 | 174 | 40.2 | 1219 |
| **3B-Digest-J3** | <LOD | 525 | <LOD | 120.3 | 10591 | <LOD | <LOD | 5.3 | 35.1 | 3384 | 822 | <LOD | 1216 | 95.1 | 611 | 12.2 | 497 | 33.4 | 1096 |
| **3B-Digest-K1** | <LOD | 3532 | <LOD | 129.8 | 11582 | <LOD | 7.3 | 18.1 | 29.8 | 16924 | 1491 | <LOD | 1788 | 204.1 | 779 | 22.1 | 450 | 53.5 | 1255 |
| **3B-Digest-K2** | <LOD | 1299 | <LOD | 72.4 | 7018 | <LOD | <LOD | 8.9 | 17.2 | 10487 | 775 | <LOD | 1299 | 174.0 | 579 | 14.6 | 140 | 28.4 | 1101 |
| **3B-Digest-K3** | <LOD | 2262 | <LOD | 1153.9 | 32988 | <LOD | 8.5 | 50.4 | 114.8 | 8303 | 1484 | <LOD | 2401 | 191.7 | 1096 | 21.7 | 15271 | 163.6 | 6300 |
| **3C-Digest-H1** | <LOD | 2393 | <LOD | 61.3 | 1325 | <LOD | 4.6 | 18.9 | 54.4 | 28378 | 518 | <LOD | 470 | 91.1 | 451 | 14.5 | 126 | 8.1 | 1281 |
| **3C-Digest-H2** | 0.16 | 1385 | <LOD | 39.6 | 1302 | <LOD | <LOD | 14.9 | 45.6 | 54536 | 412 | <LOD | 379 | 95.7 | 330 | 17.0 | 83 | 7.1 | 1058 |
| **3C-Digest-H3** | <LOD | 1149 | <LOD | 40.1 | 1150 | <LOD | <LOD | 21.5 | 57.3 | 21108 | 344 | <LOD | 380 | 85.5 | 250 | 11.5 | 108 | 6.3 | 1227 |
| **3C-Digest-J1** | <LOD | 947 | <LOD | 78.1 | 2049 | <LOD | <LOD | 11.1 | 104.9 | 7289 | 409 | <LOD | 541 | 66.3 | 300 | 13.7 | 1280 | 11.5 | 2836 |
| **3C-Digest-J2** | <LOD | 2712 | 17.2 | 75.7 | 2404 | <LOD | <LOD | 9.7 | 125.8 | 10355 | 487 | <LOD | 565 | 67.4 | 418 | 14.1 | 1744 | 12.5 | 5726 |
| **3C-Digest-J3** | 0.19 | 1081 | <LOD | 83.6 | 2313 | <LOD | <LOD | 8.8 | 110.6 | 11349 | 346 | <LOD | 513 | 122.7 | 380 | 12.3 | 1845 | 10.9 | 5731 |
| **3C-Digest-K1** | <LOD | 163 | 12.7 | 21.6 | 1335 | <LOD | <LOD | 2.9 | 52.7 | 3632 | 229 | <LOD | 324 | 46.2 | 346 | 5.5 | 1105 | 5.8 | 4936 |
| **3C-Digest-K2** | <LOD | 332 | <LOD | 15.5 | 753 | <LOD | <LOD | 2.2 | 32.7 | 1255 | 166 | <LOD | 247 | 16.4 | 284 | <LOD | 917 | 4.1 | 3936 |
| **3C-Digest-K3** | <LOD | 254 | <LOD | 9.9 | 613 | <LOD | <LOD | 1.8 | 31.8 | 1057 | 135 | <LOD | 220 | 10.7 | 231 | <LOD | 736 | 3.5 | 2835 |
| **3D-Digest-H1** | <LOD | 2609 | <LOD | 82.7 | 3499 | <LOD | <LOD | 19.4 | 47.2 | 37651 | 743 | <LOD | 787 | 127.4 | 530 | 24.8 | 155 | 17.9 | 977 |
| **3D-Digest-H2** | <LOD | 1674 | 13.6 | 34.6 | 1908 | <LOD | <LOD | 12.0 | 34.0 | 21018 | 469 | <LOD | 588 | 70.7 | 455 | 17.2 | 78 | 10.0 | 1437 |
| **3D-Digest-H3** | <LOD | 1163 | <LOD | 34.2 | 1905 | <LOD | <LOD | 10.0 | 17.1 | 41413 | 343 | <LOD | 492 | 84.4 | 381 | 20.4 | 99 | 9.9 | 898 |
| **3D-Digest-J1** | <LOD | 2218 | 19.7 | 34.1 | 2202 | <LOD | <LOD | 10.3 | 23.0 | 28708 | 535 | <LOD | 645 | 77.2 | 512 | 15.1 | 92 | 11.3 | 557 |
| **3D-Digest-J2** | <LOD | 166 | <LOD | 7.9 | 1419 | <LOD | <LOD | 5.2 | 24.6 | 13096 | 205 | <LOD | 402 | 41.1 | 378 | 10.2 | 50 | 7.0 | 895 |
| **3D-Digest-J3** | <LOD | 329 | <LOD | 14.3 | 1843 | <LOD | <LOD | 5.3 | 13.3 | 20724 | 256 | <LOD | 485 | 51.6 | 411 | 9.6 | 44 | 8.4 | 433 |
| **3D-Digest-K1** | <LOD | 1150 | <LOD | 71.4 | 1895 | <LOD | <LOD | 7.7 | 35.3 | 15108 | 327 | <LOD | 527 | 56.0 | 385 | 11.3 | 63 | 9.2 | 403 |
| **3D-Digest-K2** | <LOD | 13 | <LOD | 9.2 | 1394 | <LOD | <LOD | 2.4 | 8.2 | 4659 | 161 | <LOD | 407 | 28.1 | 404 | 7.6 | 27 | 6.0 | 296 |
| **3D-Digest-K3** | <LOD | 575 | <LOD | 28.0 | 2728 | <LOD | <LOD | 6.9 | 17.0 | 20012 | 328 | <LOD | 698 | 72.9 | 455 | 15.3 | 72 | 13.3 | 892 |

**Table A.11: Hadleigh Marsh trial pit 4 wood inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Limit of detection** | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.72 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.60 | 1.11 | 0.05 | 0.17 |
| **4D-Digest-K1** | <LOD | 10998 | <LOD | 103.8 | 9357 | <LOD | 3.8 | 4.3 | 141.1 | 11721 | 857 | 3.4 | 1102 | 142.5 | 139 | 18.5 | 141 | 38.4 | 1392 |
| **4D-Digest-K2** | <LOD | 20390 | <LOD | 107.4 | 7293 | <LOD | 4.7 | 8.6 | 79.7 | 12570 | 559 | 4.7 | 1136 | 214.9 | 191 | 20.2 | 148 | 39.6 | 1283 |
| **4D-Digest-K3** | <LOD | 15882 | <LOD | 73.1 | 6493 | <LOD | 6.5 | 23.7 | 108.4 | 16193 | 624 | 3.0 | 1062 | 183.7 | 213 | 32.1 | 308 | 33.3 | 3218 |
| **4D-Digest-L1** | <LOD | 586 | <LOD | 32.6 | 8619 | <LOD | <LOD | 2.9 | 27.7 | 6225 | 840 | <LOD | 758 | 65.3 | 158 | 14.2 | 39 | 33.5 | 884 |
| **4D-Digest-L2** | <LOD | 348 | <LOD | 28.9 | 6767 | <LOD | <LOD | <LOD | 22.5 | 4528 | 622 | <LOD | 617 | 29.7 | 75 | 8.3 | 40 | 26.2 | 457 |
| **4D-Digest-L3** | <LOD | 698 | <LOD | 35.2 | 8975 | <LOD | <LOD | 2.1 | 24.0 | 5698 | 832 | <LOD | 788 | 48.0 | 131 | 11.1 | 38 | 35.7 | 510 |
| **4D-Digest-M1** | <LOD | 78 | <LOD | 15.1 | 6534 | <LOD | <LOD | <LOD | 9.9 | 4048 | 626 | <LOD | 513 | 28.8 | 129 | 9.1 | 17 | 25.3 | 357 |
| **4D-Digest-M2** | <LOD | 59 | <LOD | 10.0 | 4163 | <LOD | <LOD | <LOD | 5.9 | 2277 | 343 | <LOD | 350 | 10.2 | 82 | 4.9 | 11 | 14.9 | 268 |
| **4D-Digest-M3** | <LOD | 295 | <LOD | 16.2 | 6659 | <LOD | <LOD | 2.3 | 13.1 | 4306 | 486 | <LOD | 539 | 26.7 | 91 | 5.9 | 25 | 21.9 | 280 |

**Table A.12: Hadleigh Marsh trial pit 3 paper inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Limit of detection** | 0.006 | 6.66 | 1.79 | 0.08 | 2.62 | 0.005 | 0.001 | 0.007 | 1.73 | 3.81 | 31 | 1.13 | 0.04 | 0.22 | 4.25 | 0.02 | 0.009 | 0.04 | 5.22 |
| **3A-Digest-E1** | 0.56 | 38447 | <LOD | 40.5 | 1519 | 0.77 | 2.19 | 14.4 | 87.8 | 2022 | 756 | 22.4 | 733 | 17.0 | 280 | 4.9 | 23.6 | 26.4 | 127 |
| **3A-Digest-E2** | 0.41 | 62037 | 7.5 | 32.8 | 1131 | 0.47 | 1.63 | 12.5 | 89.6 | 1584 | 648 | 16.8 | 528 | 14.7 | 224 | 3.9 | 26.7 | 21.6 | 107 |
| **3A-Digest-E3** | Insufficient sample for ICP-MS analysis | | | | | | | | | | | | | | | | | | |
| **3A-Digest-F1** | <LOD | 27749 | 153.0 | 14.0 | 2273 | 0.11 | 1.54 | 28.2 | 17.8 | 128747 | 309 | 1.8 | 382 | 2.5 | 323 | 6.3 | 49.0 | 6.0 | 1575 |
| **3A-Digest-F2** | 0.21 | 14723 | 99.8 | 95.8 | 4943 | 0.50 | 2.95 | 26.8 | 33.3 | 83610 | 771 | 3.9 | 900 | 66.3 | 397 | 16.4 | 133.5 | 17.3 | 1449 |
| **3A-Digest-F3** | <LOD | 66150 | 118.6 | 5.8 | 1495 | 0.60 | 1.57 | 26.0 | 18.6 | 76474 | 203 | 3.6 | 249 | 0.6 | 235 | 11.3 | 65.2 | 6.4 | 4700 |
| **3A-Digest-G1** | 0.44 | 35041 | 27.8 | 108.7 | 5380 | 0.87 | 4.56 | 20.4 | 107.2 | 7085 | 1018 | 24.2 | 1558 | 93.4 | 364 | 12.8 | 102.8 | 44.4 | 286 |
| **3A-Digest-G2** | 0.23 | 43658 | <LOD | 49.3 | 1298 | 0.50 | 1.51 | 12.3 | 80.9 | 908 | 688 | 19.3 | 534 | 10.8 | 205 | 4.1 | 22.8 | 24.0 | 73 |
| **3A-Digest-G3** | 0.27 | 25906 | <LOD | 37.3 | 1151 | 0.20 | 1.49 | 14.3 | 62.2 | 777 | 612 | 19.9 | 578 | 10.2 | 302 | 3.2 | 18.9 | 20.0 | 61 |
| **3C-Digest-E1** | 0.03 | 27745 | <LOD | 18.8 | 3021 | 2.15 | 0.63 | 6.0 | 10.5 | 1014 | 310 | 2.5 | 846 | 12.8 | 416 | 4.9 | 21.9 | 27.3 | 171 |
| **3C-Digest-E2** | 0.05 | 39574 | <LOD | 16.8 | 3061 | 0.24 | 0.56 | 4.6 | 13.1 | 1549 | 365 | 3.7 | 909 | 17.3 | 476 | 3.6 | 21.7 | 17.0 | 125 |
| **3C-Digest-E3** | 0.00 | 55346 | <LOD | 16.6 | 2473 | 0.43 | 0.18 | 3.4 | 5.4 | 390 | 205 | 2.1 | 675 | 4.1 | 308 | 2.9 | 8.3 | 20.5 | 131 |
| **3C-Digest-F1** | 0.06 | 28719 | 14.9 | 15.0 | 2516 | 0.17 | 0.33 | 5.2 | 7.2 | 805 | 328 | 3.3 | 761 | 5.4 | 467 | 2.6 | 17.7 | 10.6 | 42 |
| **3C-Digest-F2** | 0.03 | 19008 | <LOD | 10.9 | 2022 | 0.21 | 0.24 | 4.5 | 9.0 | 472 | 250 | 2.3 | 639 | 4.5 | 366 | 2.6 | 18.6 | 7.5 | 35 |
| **3C-Digest-F3** | 0.10 | 54672 | 11.1 | 12.9 | 2376 | 0.27 | 0.29 | 4.1 | 9.5 | 934 | 315 | 4.7 | 689 | 6.5 | 412 | 2.4 | 11.3 | 14.8 | 67 |
| **3C-Digest-G1** | 0.11 | 22206 | 51.3 | 81.4 | 19639 | 0.44 | 1.80 | 6.8 | 46.0 | 7152 | 812 | 7.4 | 1790 | 82.6 | 913 | 7.5 | 61.9 | 41.8 | 321 |
| **3C-Digest-G2** | 0.09 | 18315 | 20.9 | 24.3 | 3298 | 0.24 | 1.30 | 6.9 | 43.7 | 5121 | 338 | 2.8 | 938 | 43.2 | 595 | 5.7 | 39.7 | 31.6 | 746 |
| **3C-Digest-G3** | 0.13 | 12810 | <LOD | 12.2 | 3496 | 0.16 | 0.57 | 6.0 | 10.1 | 1565 | 243 | 1.6 | 839 | 12.4 | 484 | 3.7 | 29.5 | 6.8 | 57 |
| **3D-Digest-E1** | 0.06 | 899 | 3.2 | 24.1 | 4815 | 1.89 | 2.16 | 4.1 | 16.5 | 3478 | 573 | 3.0 | 1330 | 38.5 | 666 | 8.2 | 34.4 | 17.0 | 2274 |
| **3D-Digest-E2** | <LOD | 339 | 45.4 | 10.3 | 5205 | 1.26 | 0.68 | 1.2 | 12.9 | 2550 | 576 | 2.3 | 1450 | 30.4 | 822 | 3.6 | 17.9 | 17.8 | 2735 |
| **3D-Digest-E3** | 0.09 | 840 | <LOD | 18.7 | 4668 | 1.00 | 1.36 | 2.4 | 18.4 | 2917 | 566 | 3.5 | 1309 | 37.6 | 640 | 5.9 | 31.0 | 16.5 | 2804 |
| **3D-Digest-F1** | 0.04 | 4199 | 11.6 | 7.0 | 2744 | 0.20 | 1.04 | 2.2 | 6.5 | 1062 | 279 | <LOD | 733 | 21.9 | 390 | 4.1 | 10.9 | 18.3 | 1104 |
| **3D-Digest-F2** | 0.10 | 14180 | 21.5 | 5.1 | 3091 | <LOD | 0.36 | 2.1 | 2.5 | 925 | 220 | 1.8 | 674 | 2.2 | 399 | 3.0 | 4.5 | 40.9 | 502 |
| **3D-Digest-F3** | 0.01 | 11024 | 21.3 | 16.8 | 3633 | 0.03 | 1.62 | 4.2 | 9.5 | 4263 | 387 | 3.7 | 919 | 29.6 | 538 | 8.0 | 22.7 | 37.8 | 771 |
| **3D-Digest-G1** | <LOD | 5196 | <LOD | 5.3 | 3091 | <LOD | 0.25 | 1.0 | 5.1 | 1612 | 273 | 1.5 | 875 | 3.9 | 512 | 1.9 | 5.7 | 21.7 | 536 |
| **3D-Digest-G2** | 0.25 | 12843 | <LOD | 9.1 | 3613 | <LOD | 0.33 | 1.7 | 2.0 | 930 | 278 | 2.0 | 893 | 2.3 | 565 | 2.5 | 4.0 | 56.7 | 733 |
| **3D-Digest-G3** | 0.05 | 13079 | <LOD | 4.1 | 2808 | 0.13 | 0.33 | 1.6 | 1.7 | 590 | 191 | 2.1 | 608 | 1.7 | 326 | 2.8 | 7.0 | 42.5 | 392 |

**Table A.13: Hadleigh Marsh trial pit 4 paper inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Limit of detection** | 0.006 | 6.66 | 1.79 | 0.08 | 2.62 | 0.005 | 0.001 | 0.007 | 1.73 | 3.81 | 31 | 1.13 | 0.04 | 0.22 | 4.25 | 0.02 | 0.009 | 0.04 | 5.22 |
| **4A-Digest-D1** | 0.21 | 3878 | <LOD | 36.6 | 4656 | 0.18 | 1.91 | 6.9 | 27.5 | 4128 | 444 | 3.9 | 652 | 44.0 | 267 | 5.1 | 14.6 | 40.1 | 202 |
| **4A-Digest-D2** | 0.07 | 4397 | 21.0 | 52.1 | 15354 | 0.22 | 2.31 | 56.1 | 85.2 | 17434 | 681 | 8.7 | 1968 | 294.4 | 179 | 19.1 | 35.4 | 28.6 | 965 |
| **4A-Digest-D3** | 0.85 | 6342 | 9.7 | 61.5 | 5039 | 0.17 | 2.88 | 90.1 | 88.4 | 11220 | 833 | 11.8 | 2908 | 153.3 | 248 | 22.8 | 22.8 | 34.4 | 894 |
| **4A-Digest-E1** | 0.15 | 2260 | <LOD | 24.0 | 3782 | 0.15 | 1.17 | 1.8 | 8.0 | 4340 | 361 | 2.6 | 586 | 16.8 | 285 | 3.0 | 7.2 | 13.9 | 67 |
| **4A-Digest-E2** | 0.01 | 2154 | <LOD | 27.9 | 4017 | 0.11 | 1.20 | 2.1 | 6.2 | 2868 | 378 | 2.7 | 569 | 22.2 | 259 | 3.0 | 9.4 | 14.2 | 72 |
| **4A-Digest-E3** | 0.17 | 1989 | 7.8 | 10.4 | 3840 | 0.64 | 1.45 | 13.4 | 28.5 | 2912 | 407 | 4.3 | 809 | 30.1 | 215 | 8.7 | 13.4 | 14.3 | 640 |
| **4A-Digest-F1** | 0.20 | 2011 | 3.7 | 11.2 | 5852 | 0.01 | 1.05 | 1.4 | 5.7 | 2376 | 427 | 4.4 | 750 | 23.5 | 385 | 6.2 | 4.4 | 18.1 | 427 |
| **4A-Digest-F2** | 0.13 | 2581 | 4.6 | 13.1 | 4456 | 0.27 | 1.09 | 2.7 | 6.5 | 2343 | 404 | 2.1 | 618 | 31.2 | 272 | 2.5 | 4.5 | 14.8 | 217 |
| **4A-Digest-F3** | 0.16 | 1799 | <LOD | 9.4 | 3502 | 0.11 | 1.28 | 4.4 | 11.6 | 2337 | 316 | 1.9 | 505 | 45.5 | 179 | 1.5 | 7.7 | 11.0 | 407 |
| **4C-Digest-D1** | 0.29 | 22334 | 36.3 | 175.0 | 8148 | 0.91 | 2.24 | 18.6 | 15.0 | 6144 | 483 | 3.4 | 611 | 103.2 | 107 | 20.7 | 66.5 | 244.1 | 144 |
| **4C-Digest-D2** | <LOD | 19678 | 56.5 | 79.4 | 7694 | 0.16 | 0.55 | 9.3 | 5.2 | 2458 | 340 | 2.2 | 396 | 9.2 | 127 | 6.1 | 18.6 | 204.1 | 33 |
| **4C-Digest-D3** | <LOD | 30636 | 52.4 | 43.8 | 9104 | 0.06 | 0.59 | 12.9 | 3.9 | 1740 | 238 | 2.8 | 302 | 16.1 | 91 | 4.6 | 17.7 | 325.2 | 24 |
| **4C-Digest-E1** | 0.34 | 5784 | <LOD | 75.5 | 5525 | 1.08 | 0.93 | 22.2 | 28.0 | 2530 | 638 | 1.6 | 702 | 20.7 | 132 | 9.6 | 26.2 | 64.0 | 283 |
| **4C-Digest-E2** | 0.10 | 8982 | 39.7 | 30.0 | 5268 | 0.51 | 0.83 | 27.5 | 21.5 | 2412 | 460 | 4.2 | 752 | 16.7 | 124 | 6.0 | 18.7 | 80.1 | 37 |
| **4C-Digest-E3** | 0.08 | 7557 | <LOD | 41.2 | 5104 | 0.38 | 1.60 | 29.1 | 496.6 | 6146 | 592 | 4.9 | 1005 | 57.2 | 81 | 10.2 | 67.4 | 64.0 | 147 |
| **4C-Digest-F1** | 0.07 | 10577 | 53.9 | 108.0 | 5814 | 0.07 | 0.92 | 7.0 | 12.9 | 2652 | 374 | 2.5 | 530 | 15.6 | 117 | 6.0 | 16.3 | 167.5 | 35 |
| **4C-Digest-F2** | 0.01 | 15531 | 39.8 | 118.2 | 6670 | 0.12 | 0.98 | 13.4 | 7.4 | 2281 | 423 | 5.0 | 515 | 16.0 | 98 | 5.2 | 30.0 | 197.5 | 41 |
| **4C-Digest-F3** | 0.03 | 8656 | 22.2 | 96.3 | 4589 | 0.08 | 0.81 | 10.0 | 9.1 | 1910 | 330 | 4.5 | 447 | 10.7 | 94 | 4.1 | 14.9 | 98.3 | 21 |
| **4D-Digest-D1** | 0.77 | 1011 | <LOD | 29.3 | 6533 | 0.72 | 1.22 | 17.6 | 14.2 | 5407 | 516 | 2.2 | 830 | 47.7 | 195 | 7.5 | 15.5 | 22.2 | 139 |
| **4D-Digest-D2** | 0.12 | 383 | 2.0 | 19.6 | 5355 | 1.26 | 0.67 | 13.0 | 12.6 | 4512 | 414 | 2.2 | 634 | 64.8 | 163 | 3.2 | 9.5 | 15.8 | 160 |
| **4D-Digest-D3** | 0.09 | 238 | 12.4 | 23.5 | 4026 | 0.10 | 0.38 | 23.6 | 6.4 | 7710 | 317 | 1.4 | 508 | 14.4 | 135 | 2.0 | 8.4 | 13.7 | 46 |
| **4D-Digest-E1** | 0.25 | 1737 | <LOD | 28.8 | 6061 | 5.00 | 1.00 | 20.9 | 21.4 | 5178 | 464 | 2.4 | 743 | 74.2 | 138 | 10.2 | 21.4 | 20.7 | 168 |
| **4D-Digest-E2** | 0.05 | 305 | 17.7 | 18.9 | 5151 | 0.84 | 0.34 | 14.3 | 8.5 | 1617 | 476 | 2.0 | 664 | 23.3 | 168 | 2.0 | 6.7 | 16.2 | 67 |
| **4D-Digest-E3** | 0.04 | 402 | 32.5 | 22.9 | 5570 | 0.25 | 0.36 | 8.0 | 10.2 | 2098 | 508 | 3.1 | 691 | 29.5 | 170 | 2.3 | 14.4 | 18.9 | 76 |
| **4D-Digest-F1** | 0.41 | 2785 | <LOD | 74.3 | 7987 | 14.02 | 2.36 | 28.2 | 37.8 | 9991 | 824 | 4.2 | 1074 | 80.0 | 172 | 24.6 | 734.3 | 29.4 | 316 |
| **4D-Digest-F2** | 0.16 | 1009 | 11.3 | 22.2 | 6421 | 2.33 | 1.25 | 13.6 | 21.0 | 7800 | 438 | 2.5 | 652 | 61.7 | 132 | 12.5 | 18.3 | 22.7 | 133 |
| **4D-Digest-F3** | 0.25 | 2480 | 33.1 | 74.5 | 7391 | 12.83 | 1.52 | 22.6 | 40.7 | 7152 | 720 | 3.5 | 970 | 68.7 | 184 | 12.7 | 84.4 | 30.4 | 227 |

**Table A.14: Hadleigh Marsh trial pit 3 textiles inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Limit of detection | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.72 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.6 | 1.11 | 0.05 | 0.17 |
| 3B-Digest-E1 | <LOD | 5376 | 21.2 | 171 | 88938 | <LOD | 85.5 | 27.2 | 41 | 11744 | 2351 | <LOD | 1881 | 235 | 607 | 41.0 | 353 | 343 | 816 |
| 3B-Digest-E2 | <LOD | 3311 | 56.3 | 118 | 147027 | <LOD | 412.1 | 26.3 | 96 | 6386 | 1937 | <LOD | 2276 | 174 | 639 | 123.5 | 236 | 731 | 889 |
| 3B-Digest-E3 | <LOD | 5555 | <LOD | 228 | 90704 | <LOD | 420.6 | 39.3 | 48 | 17377 | 2524 | <LOD | 2083 | 273 | 707 | 142.5 | 371 | 391 | 739 |
| 3B-Digest-F1 | <LOD | 4202 | 33.9 | 141 | 111833 | <LOD | 29.2 | 19.0 | 37 | 9572 | 2242 | <LOD | 1804 | 244 | 590 | 23.4 | 291 | 358 | 685 |
| 3B-Digest-F2 | <LOD | 6860 | 17.3 | 229 | 94052 | <LOD | 38.7 | 25.3 | 90 | 18170 | 2870 | <LOD | 2158 | 307 | 859 | 29.7 | 459 | 286 | 815 |
| 3B-Digest-F3 | <LOD | 4419 | 27.9 | 140 | 102100 | <LOD | 19.2 | 17.0 | 25 | 9384 | 2177 | <LOD | 1694 | 245 | 539 | 15.2 | 275 | 317 | 520 |
| 3B-Digest-G1 | <LOD | 2667 | <LOD | 81 | 133014 | <LOD | <LOD | 8.6 | 20 | 5719 | 1692 | <LOD | 1547 | 174 | 494 | 8.8 | 177 | 440 | 534 |
| 3B-Digest-G2 | <LOD | 4051 | <LOD | 88 | 120392 | <LOD | 7.0 | 11.6 | 26 | 9189 | 2077 | <LOD | 1706 | 205 | 555 | 10.5 | 199 | 356 | 566 |
| 3B-Digest-G3 | <LOD | 3961 | 12.5 | 84 | 137296 | <LOD | 6.2 | 10.6 | 20 | 7656 | 1969 | <LOD | 1614 | 179 | 504 | 10.0 | 174 | 362 | 639 |

**Table A.15: Hadleigh Marsh trial pit 4 textiles inorganic contaminant concentrations**

| All data mg kg$^{-1}$ | Ag | Al | B | Ba | Ca | Cd | Co | Cr | Cu | Fe | K | Li | Mg | Mn | Na | Ni | Pb | Sr | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Limit of detection | 0.06 | 0.12 | 5.83 | 0.09 | 0.16 | 0.72 | 0.31 | 0.12 | 0.08 | 6.55 | 0.17 | 0.03 | 0.09 | 0.07 | 0.07 | 0.6 | 1.11 | 0.05 | 0.17 |
| 4B-Digest-D1 | 1.24 | 877 | <LOD | 424 | 3762 | <LOD | <LOD | 74.8 | 185 | 9698 | 301 | <LOD | 942 | 54 | 185 | 34.9 | 1016 | 33 | 238 |
| 4B-Digest-D2 | 0.67 | 8569 | <LOD | 126 | 12905 | <LOD | <LOD | 449.0 | 254 | 23122 | 664 | 16.8 | 753 | 238 | 193 | 26.9 | 288 | 62 | 699 |
| 4B-Digest-D3 | 0.58 | 898 | <LOD | 84 | 8828 | <LOD | <LOD | 875.1 | 218 | 33646 | 286 | <LOD | 482 | 102 | 105 | 91.9 | 565 | 38 | 597 |
| 4B-Digest-E1 | 0.95 | 15212 | <LOD | 125 | 23759 | <LOD | <LOD | 92.3 | 313 | 32374 | 1190 | 30.9 | 1463 | 279 | 758 | 17.3 | 53 | 101 | 690 |
| 4B-Digest-E2 | 1.14 | 11972 | <LOD | 77 | 22104 | <LOD | <LOD | 123.4 | 106 | 29395 | 837 | 25.9 | 1131 | 230 | 364 | 16.5 | 46 | 75 | 335 |
| 4B-Digest-E3 | 0.67 | 16378 | <LOD | 87 | 25640 | <LOD | <LOD | 108.7 | 98 | 36098 | 1222 | 35.4 | 1249 | 344 | 389 | 19.5 | 39 | 102 | 527 |
| 4B-Digest-F1 | 0.60 | 8453 | <LOD | 94 | 19150 | <LOD | <LOD | 64.0 | 99 | 22454 | 662 | 22.8 | 967 | 389 | 264 | 21.4 | 82 | 103 | 426 |
| 4B-Digest-F2 | 0.67 | 1486 | <LOD | 73 | 14717 | <LOD | <LOD | 197.1 | 1767 | 112808 | 500 | <LOD | 620 | 471 | 156 | 114.0 | 196 | 47 | 573 |
| 4B-Digest-F3 | 1.28 | 22331 | 13.6 | 166 | 14855 | <LOD | <LOD | 51.7 | 83 | 28137 | 1032 | 23.7 | 1439 | 318 | 365 | 20.5 | 47 | 250 | 522 |
| 4D-Digest-G1 | <LOD | 4298 | 20.3 | 61 | 25735 | <LOD | <LOD | 7.6 | 109 | 12648 | 601 | <LOD | 754 | 320 | 172 | 8.4 | 44 | 100 | 91 |
| 4D-Digest-G2 | 0.57 | 1798 | <LOD | 51 | 31507 | <LOD | <LOD | 14.0 | 126 | 17563 | 676 | <LOD | 897 | 406 | 209 | 14.3 | 97 | 99 | 128 |
| 4D-Digest-G3 | 0.70 | 1614 | <LOD | 50 | 37087 | <LOD | <LOD | 11.5 | 120 | 18125 | 524 | <LOD | 908 | 515 | 136 | 15.0 | 67 | 124 | 110 |
| 4D-Digest-H1 | 0.49 | 1303 | <LOD | 45 | 57978 | <LOD | <LOD | 5.3 | 101 | 19981 | 489 | <LOD | 1100 | 729 | 239 | 8.8 | 35 | 182 | 89 |
| 4D-Digest-H2 | 0.48 | 966 | <LOD | 41 | 45576 | <LOD | <LOD | 4.8 | 102 | 13847 | 421 | <LOD | 888 | 537 | 197 | <LOD | 44 | 130 | 68 |
| 4D-Digest-H3 | 0.89 | 795 | <LOD | 39 | 40187 | <LOD | <LOD | 4.9 | 104 | 13784 | 377 | <LOD | 846 | 459 | 158 | 5.2 | 34 | 123 | 65 |
| 4D-Digest-J1 | <LOD | 79403 | <LOD | 80 | 12720 | <LOD | <LOD | 8.9 | 121 | 9567 | 429 | <LOD | 551 | 132 | 43 | 8.4 | 80 | 37 | 75 |
| 4D-Digest-J2 | 1.14 | 2382 | <LOD | 106 | 12260 | <LOD | <LOD | 18.9 | 129 | 9692 | 581 | <LOD | 663 | 239 | 103 | 8.2 | 199 | 49 | 101 |
| 4D-Digest-J3 | 0.42 | 778 | <LOD | 50 | 25403 | <LOD | <LOD | 5.2 | 117 | 8189 | 382 | <LOD | 635 | 274 | 109 | <LOD | 48 | 75 | 58 |

**Table A.16: Leigh Marshes trial pit 5 matrix material organic contaminant concentrations**

| Data µg kg$^{-1}$, except dry solids (%) | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | cyclopenta(cd)pyrene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene | dry Solids (%) at 30°C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **5E - MATRIX A** | 39 | 162 | 413 | 276 | 3220 | 1730 | 1730 | 2330 | 1310 | 1370 | 891 | 1830 | 296 | <10 | 403 | 3070 | 101 | 1850 | 642 | 444 | 1340 | 2790 | 66.7 |
| **5E - MATRIX B** | 60 | 213 | 441 | 627 | 4060 | 2310 | 2050 | 2940 | 1600 | 1600 | 1120 | 2300 | 345 | <10 | 481 | 5180 | 211 | 2190 | 694 | 557 | 2720 | 4080 | 65.3 |
| **5E - MATRIX C** | 26 | 50 | 245 | 162 | 2050 | 1200 | 1210 | 1570 | 921 | 974 | 480 | 1220 | 233 | <10 | 276 | 2070 | 51 | 1130 | 403 | 281 | 913 | 1860 | 70.9 |
| **5F - MATRIX A** | 88 | 58 | 209 | 352 | 1520 | 887 | 696 | 1070 | 660 | 672 | 294 | 1200 | 161 | <10 | 248 | 2030 | 140 | 690 | 1080 | 140 | 2220 | 1660 | 64.3 |
| **5F - MATRIX B** | 79 | 116 | 374 | 533 | 3550 | 2030 | 2070 | 2880 | 1650 | 1540 | 607 | 2050 | 408 | <10 | 455 | 3480 | 183 | 1560 | 767 | 383 | 2290 | 3160 | 72.1 |
| **5F - MATRIX C** | 237 | 44 | 674 | 1130 | 6050 | 3550 | 3510 | 4560 | 2180 | 1990 | 1490 | 353 | 468 | <10 | 558 | 8690 | 312 | 2590 | 483 | 417 | 5180 | 6950 | 65.1 |
| **5G - MATRIX A** | 67 | 281 | 580 | 504 | 4110 | 2050 | 2310 | 3000 | 1640 | 1790 | 1120 | 2030 | 360 | <10 | 441 | 4120 | 115 | 1920 | 1880 | 618 | 2760 | 3650 | 67.8 |
| **5G - MATRIX B** | 111 | 328 | 862 | 1290 | 5700 | 3430 | 3120 | 3720 | 2030 | 2150 | 1980 | 3290 | 560 | <10 | 652 | 8290 | 372 | 2630 | 856 | 871 | 5320 | 6700 | 69.2 |
| **5G - MATRIX C** | 30 | 136 | 376 | 224 | 2890 | 1180 | 1330 | 2120 | 1350 | 1180 | <20 | 1260 | 202 | 111 | 445 | 1840 | 70 | 1350 | 335 | 296 | 1160 | 1940 | 69.5 |
| **5H - MATRIX A** | 81 | 877 | 1020 | 1290 | 10500 | 5840 | 8250 | 7700 | 4530 | 4490 | 2780 | 5960 | 983 | 813 | 1240 | 7830 | 270 | 5890 | 2580 | 1760 | 4990 | 11000 | 70.7 |
| **5H - MATRIX B** | 84 | 1100 | 1230 | 1450 | 10400 | 6850 | 8530 | 8110 | 5130 | 5350 | 2550 | 6410 | 1050 | 915 | 1480 | 8390 | 241 | 6640 | 3370 | 1560 | 6960 | 12100 | 68.1 |
| **5H - MATRIX C** | 46 | 869 | 890 | 632 | 8380 | 4350 | 5990 | 6020 | 3870 | 3420 | 2210 | 4350 | 648 | 1670 | 1000 | 5390 | 160 | 4170 | 1080 | 1250 | 2720 | 8200 | 68.6 |

**Table A.17: Leigh Marshes trial pit 6 matrix material organic contaminant concentrations**

| Data µg kg$^{-1}$, except dry solids (%) | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | cyclopenta(cd)pyrene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene | dry Solids (%) at 30°C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **6E - MATRIX A** | 703 | 87 | 1480 | 3780 | 26500 | 15700 | 15700 | 19300 | 10300 | 8310 | 7190 | 24400 | 1900 | <10 | 5560 | 22200 | 1160 | 10200 | 8230 | 3220 | 10300 | 16500 | 64.4 |
| **6E - MATRIX B** | 112 | 45 | 413 | 339 | 2830 | 1470 | 1750 | 2010 | 1240 | 1360 | 815 | 1350 | 267 | <10 | 430 | 2820 | 194 | 1980 | 861 | 457 | 1290 | 2460 | 70.7 |
| **6E - MATRIX C** | 311 | 58 | 667 | 1140 | 6660 | 4320 | 3390 | 4990 | 2590 | 2290 | 1680 | 4680 | 629 | <10 | 862 | 7660 | 501 | 3430 | 2090 | 889 | 3670 | 5580 | 67.4 |
| **6F - MATRIX A** | 363 | 86 | 520 | 718 | 3200 | 1830 | 1780 | 2290 | 1270 | 1400 | 919 | 2460 | 187 | <10 | 372 | 5820 | 573 | 1850 | 1950 | 475 | 3340 | 4860 | 59.3 |
| **6F - MATRIX B** | 400 | 92 | 446 | 551 | 2980 | 1610 | 1570 | 2180 | 1380 | 1220 | 721 | 1750 | 244 | <10 | 434 | 3630 | 507 | 1710 | 1800 | 456 | 2980 | 3300 | 61.2 |
| **6F - MATRIX C** | 332 | 59 | <500 | 584 | 2700 | 1510 | 1620 | 2060 | 1260 | 1260 | 634 | 1650 | 255 | 357 | 367 | 3700 | 423 | 1720 | 3140 | 433 | 2630 | 3240 | 66.8 |
| **6G - MATRIX A** | 367 | 107 | 373 | 644 | 2270 | 1070 | 1280 | 1710 | 904 | 1030 | 534 | 1710 | 297 | <10 | 1030 | 2830 | 518 | 1340 | 3570 | 294 | 3060 | 2660 | 59.6 |
| **6G - MATRIX B** | 302 | 78 | 490 | 544 | 3450 | 1650 | 2090 | 2630 | 817 | 1620 | 817 | 2030 | 423 | <10 | 453 | 3730 | 349 | 1720 | 2740 | 392 | 2570 | 3250 | 66 |
| **6G - MATRIX C** | 345 | 208 | 467 | 647 | 3960 | 2040 | 2350 | 3020 | 1550 | 1510 | 940 | 2470 | 392 | <10 | 448 | 4690 | 464 | 1650 | 2590 | 556 | 2780 | 4350 | 69.7 |
| **6H - MATRIX A** | 1020 | 74 | 592 | 742 | 5340 | 2660 | 2880 | 3860 | 2010 | 1970 | 1490 | 2930 | 495 | <10 | 389 | 6650 | 838 | 1960 | 3560 | 709 | 4950 | 709 | 78.7 |
| **6H - MATRIX B** | 171 | 98 | 556 | 531 | 4350 | 1990 | 2420 | 3070 | 1690 | 1850 | 1270 | 2580 | 471 | <10 | 461 | 5220 | 251 | 1930 | 1340 | 475 | 2830 | 4350 | 77.7 |
| **6H - MATRIX C** | 423 | 75 | 365 | 419 | 3080 | 1470 | 1650 | 2360 | 1240 | 1290 | 723 | 1830 | 380 | <10 | 368 | 3320 | 466 | 1220 | 4890 | 392 | 1910 | 2890 | 67.6 |

**Table A.18: Hadleigh Marsh trial pit 3 matrix material organic contaminant concentrations**

| Data µg kg$^{-1}$, except dry solids (%) | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | cyclopenta(cd)pyrene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene | dry Solids (%) at 30°C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **3E - MATRIX A** | 223 | 82 | 439 | 380 | 3800 | 1700 | 2270 | 2850 | 1460 | 1560 | 954 | 1930 | 395 | <10 | 494 | 3250 | 215 | 2540 | 114 | 472 | 1010 | 3020 | 60.6 |
| **3E - MATRIX B** | 70 | 102 | 351 | 194 | 2790 | 1200 | 1630 | 2080 | 1170 | 1240 | 2080 | 1440 | 314 | <10 | 387 | 2210 | 74 | 1970 | 87 | 456 | 1190 | 2180 | 59.5 |
| **3E - MATRIX C** | 30 | 211 | 450 | 196 | 4360 | 1540 | 2130 | 3010 | 2140 | 1950 | 1030 | 1930 | 378 | <20 | 515 | 2650 | 52 | 2930 | 93 | 321 | 859 | 2860 | 63.1 |
| **3F - MATRIX A** | 131 | 236 | 646 | 520 | 4600 | 2440 | 2820 | 3340 | 2000 | 1920 | 1250 | 2670 | 422 | <10 | 578 | 4990 | 201 | 2470 | 202 | 564 | 2240 | 4630 | 80.1 |
| **3F - MATRIX B** | 9 | 16 | <30 | 23 | 320 | 139 | 200 | 243 | 155 | 148 | 77 | 166 | 41 | <10 | 37 | 247 | <10 | 195 | 20 | 50 | 84 | 267 | 73.7 |
| **3F - MATRIX C** | 26 | 70 | 152 | 183 | 2270 | 1350 | 1310 | 1670 | 606 | 767 | 606 | 1370 | 193 | <10 | 260 | 2730 | 57 | 979 | 113 | 337 | 779 | 2230 | 72.5 |
| **3G - MATRIX A** | 173 | 81 | 192 | 640 | 2170 | 1420 | 1330 | 1610 | 870 | 988 | 566 | 1330 | 208 | <10 | 242 | 4350 | 345 | 1430 | 139 | 365 | 2990 | 3610 | 65.8 |
| **3G - MATRIX B** | 117 | 134 | 293 | 407 | 2920 | 1570 | 1650 | 2120 | 1380 | 1360 | 798 | 2220 | 374 | <10 | 400 | 4880 | 182 | 2160 | 141 | 399 | 2300 | 4840 | 68.2 |
| **3G - MATRIX C** | 115 | 37 | 88 | 344 | 1490 | 810 | 802 | 1020 | 586 | 645 | 469 | 843 | 142 | <10 | 140 | 1890 | 126 | 890 | 58 | 219 | 465 | 1700 | 66.4 |
| **3H - MATRIX A** | 28 | 28 | <30 | 93 | 819 | 387 | 491 | 577 | 315 | 309 | 207 | 464 | 95 | <10 | 93 | 722 | 32 | 507 | 107 | 122 | 433 | 794 | 68.7 |
| **3H - MATRIX B** | 581 | 273 | 641 | 1020 | 6930 | 3970 | 3930 | 5810 | 2710 | 2760 | 1860 | 4290 | 720 | <10 | 957 | 9420 | 545 | 4770 | 465 | 1080 | 3670 | 8200 | 75.7 |
| **3H - MATRIX C** | 110 | 248 | 469 | 903 | 4550 | 2650 | 2590 | 3360 | 1660 | 1730 | 1190 | 2780 | 516 | <10 | 587 | 6710 | 259 | 3020 | 204 | 794 | 4300 | 5390 | 75.7 |

**Table A.19: Hadleigh Marsh trial pit 4 matrix material organic contaminant concentrations**

| Data µg kg$^{-1}$, except dry solids (%) | acenaphthene | acenaphthylene | anthanthrene | anthracene | benzo(b+k)fluoranthene | benzo(a)anthracene | benzo(a)pyrene | benzo(b)fluoranthene | benzo(e)pyrene | benzo(ghi)perylene | benzo(k)fluoranthene | chrysene | coronene | cyclopenta(cd)pyrene | dibenzo(ah)anthracene | fluoranthene | fluorene | indeno(1,2,3)cdpyrene | naphthalene | perylene | phenanthrene | pyrene | dry Solids (%) at 30°C |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **4E - MATRIX A** | 4 | 8 | 46 | 29 | 355 | 149 | 176 | 247 | 141 | 150 | 108 | 131 | 43 | 12 | 29 | 322 | 12 | 159 | 33 | 47 | 123 | 296 | 75.7 |
| **4E - MATRIX B** | 21 | 14 | <100 | 39 | 494 | 183 | 224 | 357 | 208 | 232 | 136 | 193 | 61 | 23 | 55 | 423 | 26 | 234 | 49 | <70 | 188 | 396 | 67 |
| **4E - MATRIX C** | 25 | 28 | 159 | 74 | 806 | 324 | 382 | 577 | 334 | 344 | 228 | 342 | 100 | 21 | 94 | 916 | 33 | 445 | 77 | 112 | 347 | 821 | 69.6 |
| **4G - MATRIX A** | <4 | <5 | <30 | <20 | 88 | 41 | 51 | 61 | 37 | 44 | 27 | 42 | 10 | <10 | 9 | 75 | <10 | 46 | 11 | <30 | 23 | 72 | 81.9 |
| **4G - MATRIX B** | 15 | 19 | 34 | 79 | 1380 | 675 | 719 | 1090 | 548 | 509 | 345 | 699 | 145 | <10 | 160 | 1050 | 28 | 817 | 264 | 204 | 361 | 998 | 73.1 |
| **4G - MATRIX C** | 27 | 52 | 177 | 189 | 917 | 451 | 545 | 666 | 403 | 439 | 251 | 530 | 121 | 101 | 103 | 755 | 48 | 481 | 673 | 143 | 458 | 1010 | 56.7 |
| **4H - MATRIX B** | 31 | 9 | <30 | 79 | 362 | 312 | 146 | 271 | 147 | 139 | 91 | 342 | 52 | <10 | 39 | 3550 | 99 | 198 | 72 | 33 | 1960 | 2430 | 68.7 |
| **4H - MATRIX C** | 31 | 16 | 139 | 78 | 954 | 404 | 493 | 710 | 373 | 420 | 245 | 443 | 137 | <10 | 116 | 857 | 44 | 674 | 73 | 156 | 348 | 825 | 63.9 |

# Appendix B. Full datasets for Chapter 4

**Table A.20: Field moisture in matrix material sub-samples calculated on a dry mass basis**

| Leigh Marshes trial pit 5 sample name | Moisture content (%) | Hadleigh Marsh trial pit 3 sample name | Moisture content (%) |
|---|---|---|---|
| Box 5ABCD mixed A | 71.5 | Box 3ABCD mixed-A | 59.5 |
| Box 5ABCD mixed B | 70.5 | Box 3ABCD mixed-B | 61.7 |
| Box 5ABCD mixed C | 69.9 | Box 3ABCD mixed-C | 26.3 |
| | | Box 3ABCD mixed-D | 58.9 |
| | | Box 3ABCD mixed-E | 58.5 |
| | | Box 3ABCD mixed-F | 58.3 |

**Table A.21: Moisture content in the post-leaching Leigh Marshes matrix material calculated on a dry mass basis**

| Leachate name | Moisture content (%) | Leachate name | Moisture content (%) |
|---|---|---|---|
| Trial pit 5-leach-River-1 | 156.0 | Trial pit 5-leach-Sea-1 | 181.7 |
| Trial pit 5-leach-River-2 | 211.6 | Trial pit 5-leach-Sea-2 | 161.6 |
| Trial pit 5-leach-River-3 | 156.9 | Trial pit 5-leach-Sea-3 | 152.2 |
| Trial pit 5-leach-River-4 | 187.1 | Trial pit 5-leach-Sea-4 | 150.3 |
| Trial pit 5-leach-River-5 | 192.5 | Trial pit 5-leach-Sea-5 | 149.6 |
| Trial pit 5-leach-River-6 | 156.4 | Trial pit 5-leach-Sea-6 | 150.1 |
| Trial pit 5-leach-River-7 | 156.8 | Trial pit 5-leach-Sea-7 | 155.7 |
| Trial pit 5-leach-River-8 | 213.6 | Trial pit 5-leach-Sea-8 | 154.1 |
| Trial pit 5-leach-River-9 | 194.2 | Trial pit 5-leach-Sea-9 | 149.8 |
| Trial pit 5-leach-River-10 | 242.3 | Trial pit 5-leach-Sea-10 | 175.6 |
| Trial pit 5-leach-River-11 | 213.4 | Trial pit 5-leach-Sea-11 | 151.7 |
| Trial pit 5-leach-River-12 | 213.4 | Trial pit 5-leach-Sea-12 | 161.5 |

**Table A.22: Moisture content in the post-leaching Hadleigh Marsh matrix material calculated on a dry mass basis**

| Leachate name | Moisture content (%) | Leachate name | Moisture content (%) |
|---|---|---|---|
| Trial pit 3-leach-River-1 | 149.0 | Trial pit 3-leach-Sea-1 | 162.0 |
| Trial pit 3-leach-River-2 | 159.2 | Trial pit 3-leach-Sea-2 | 157.4 |
| Trial pit 3-leach-River-3 | 164.9 | Trial pit 3-leach-Sea-3 | 161.6 |
| Trial pit 3-leach-River-4 | 152.5 | Trial pit 3-leach-Sea-4 | 144.3 |
| Trial pit 3-leach-River-5 | 160.3 | Trial pit 3-leach-Sea-5 | 159.6 |
| Trial pit 3-leach-River-6 | 162.3 | Trial pit 3-leach-Sea-6 | 154.8 |
| Trial pit 3-leach-River-7 | 162.9 | Trial pit 3-leach-Sea-7 | 166.6 |
| Trial pit 3-leach-River-8 | 155.4 | Trial pit 3-leach-Sea-8 | 143.5 |
| Trial pit 3-leach-River-9 | 172.2 | Trial pit 3-leach-Sea-9 | 121.4 |
| Trial pit 3-leach-River-10 | 154.6 | Trial pit 3-leach-Sea-10 | 144.8 |
| Trial pit 3-leach-River-11 | 169.8 | Trial pit 3-leach-Sea-11 | 132.0 |
| Trial pit 3-leach-River-12 | 179.5 | Trial pit 3-leach-Sea-12 | 144.1 |

**Table A.23: DO, redox potential, temperature and pH in the Leigh Marshes leachates**

| Time | Leachant | Sample name | DO (%) | Redox (mV) | Temp. (°C) | pH |
|---|---|---|---|---|---|---|
| Start - 0 hrs | Deionised water | Trial pit 5-leach-River-Blank 1 | 94.7 | 358 | 22.6 | 5.4 |
| | | Trial pit 5-leach-River-Blank 3 | 102.1 | 292 | 22.4 | 6.2 |
| | | Trial pit 5-leach-River-10 | 96.2 | 307 | 20.9 | 7.3 |
| | | Trial pit 5-leach-River-11 | 97.5 | 301 | 22.1 | 7.3 |
| | | Trial pit 5-leach-River-12 | 104.5 | 351 | 22.1 | 7.3 |
| | Artificial seawater | Trial pit 5-leach-Sea-Blank 1 | 99.8 | 153 | 23.2 | 8.1 |
| | | Trial pit 5-leach-Sea-Blank 3 | 98.8 | 145 | 23.1 | 8.3 |
| | | Trial pit 5-leach-Sea-10 | 101.8 | 253 | 22.5 | 7.3 |
| | | Trial pit 5-leach-Sea-11 | 102.7 | 311 | 22.3 | 7.4 |
| | | Trial pit 5-leach-Sea-12 | 100.1 | 229 | 22.6 | 7.4 |
| End - 24 hrs | Deionised water | Trial pit 5-leach-River-Blank 1 | 98.5 | 266 | 24.9 | 5.7 |
| | | Trial pit 5-leach-River-Blank 3 | 98.1 | 283 | 24.9 | 6.1 |
| | | Trial pit 5-leach-River-10 | 95.8 | 310 | 24.8 | 7.2 |
| | | Trial pit 5-leach-River-11 | 96.6 | 302 | 24.6 | 7.3 |
| | | Trial pit 5-leach-River-12 | 111.0 | 302 | 24.4 | 7.4 |
| | Artificial seawater | Trial pit 5-leach-Sea-Blank 1 | 96.2 | 126 | 24.4 | 8.2 |
| | | Trial pit 5-leach-Sea-Blank 3 | 96.4 | 185 | 24.6 | 8.2 |
| | | Trial pit 5-leach-Sea-10 | 98.5 | 182 | 23.7 | 7.2 |
| | | Trial pit 5-leach-Sea-11 | 95.0 | 220 | 24.2 | 7.1 |
| | | Trial pit 5-leach-Sea-12 | 88.2 | 255 | 24.3 | 7.1 |

**Table A.24: DO, redox potential, temperature and pH in the Hadleigh Marsh leachates**

| Time | Leachant | Sample name | DO (%) | Redox (mV) | Temp. (°C) | pH |
|---|---|---|---|---|---|---|
| Start - 0 hrs | Deionised water | Trial pit 3-leach-River-Blank 2 | 94.7 | 197 | 21.7 | 5.8 |
| | | Trial pit 3-leach-River-Blank 4 | 94.8 | 216 | 21.8 | 5.5 |
| | | Trial pit 3-leach-River-10 | 103.8 | 188 | 21.7 | 7.8 |
| | | Trial pit 3-leach-River-11 | 100.8 | 198 | 21.6 | 7.9 |
| | | Trial pit 3-leach-River-12 | 96.7 | 222 | 21.4 | 8.1 |
| | Artificial seawater | Trial pit 3-leach-Sea-Blank 2 | 89.6 | 95 | 22.3 | 8.3 |
| | | Trial pit 3-leach-Sea-Blank 4 | 87.3 | 118 | 21.8 | 8.2 |
| | | Trial pit 3-leach-Sea-10 | 89.4 | 114 | 22.3 | 7.6 |
| | | Trial pit 3-leach-Sea-11 | 96.6 | 120 | 22.1 | 7.8 |
| | | Trial pit 3-leach-Sea-12 | 97.0 | 125 | 22.0 | 7.6 |
| End - 24 hrs | Deionised water | Trial pit 3-leach-River-Blank 2 | 98.9 | 189 | 23.0 | 7.5 |
| | | Trial pit 3-leach-River-Blank 4 | 98.3 | 184 | 23.1 | 7.2 |
| | | Trial pit 3-leach-River-10 | 106.7 | 154 | 22.8 | 7.3 |
| | | Trial pit 3-leach-River-11 | 101.8 | 151 | 22.9 | 7.5 |
| | | Trial pit 3-leach-River-12 | 102.1 | 166 | 22.7 | 7.6 |
| | Artificial seawater | Trial pit 3-leach-Sea-Blank 2 | 98.5 | 127 | 24.2 | 8.0 |
| | | Trial pit 3-leach-Sea-Blank 4 | 92.3 | 123 | 24.2 | 8.1 |
| | | Trial pit 3-leach-Sea-10 | 95.8 | 146 | 24.1 | 7.3 |
| | | Trial pit 3-leach-Sea-11 | 100.9 | 162 | 24.0 | 7.1 |
| | | Trial pit 3-leach-Sea-12 | 104.3 | 158 | 23.7 | 7.2 |

**Table A.25: DOC concentrations in the leachates - Leigh Marshes**

| Leachate name | DOC (mg $L^{-1}$) | Leachate name | DOC (mg $L^{-1}$) |
|---|---|---|---|
| Trial pit 5-leach-River-1 | 5.5 | Trial pit 5-leach-Sea-1 | 17.4 |
| Trial pit 5-leach-River-2 | 14.5 | Trial pit 5-leach-Sea-2 | 18.4 |
| Trial pit 5-leach-River-3 | 14.1 | Trial pit 5-leach-Sea-3 | 18.4 |
| Trial pit 5-leach-River-4 | 15.1 | Trial pit 5-leach-Sea-4 | 17.8 |
| Trial pit 5-leach-River-5 | 15.6 | Trial pit 5-leach-Sea-5 | 19.1 |
| Trial pit 5-leach-River-6 | 15.6 | Trial pit 5-leach-Sea-6 | 18.6 |
| Trial pit 5-leach-River-7 | 14.7 | Trial pit 5-leach-Sea-7 | 17.3 |
| Trial pit 5-leach-River-8 | 14.9 | Trial pit 5-leach-Sea-8 | 18.1 |
| Trial pit 5-leach-River-9 | 14.9 | Trial pit 5-leach-Sea-9 | 18.7 |
| Trial pit 5-leach-River-10 | 16.1 | Trial pit 5-leach-Sea-10 | 36.4 |
| Trial pit 5-leach-River-11 | 7.0 | Trial pit 5-leach-Sea-11 | 19.8 |
| Trial pit 5-leach-River-12 | 14.9 | Trial pit 5-leach-Sea-12 | 20.4 |

**Table A.26: DOC concentrations in the leachates - Hadleigh Marsh**

| Leachate name | DOC (mg $L^{-1}$) | Leachate name | DOC (mg $L^{-1}$) |
|---|---|---|---|
| Trial pit 3-leach-River-1 | 25.0 | Trial pit 3-leach-Sea-1 | 41.1 |
| Trial pit 3-leach-River-2 | 37.5 | Trial pit 3-leach-Sea-2 | 41.0 |
| Trial pit 3-leach-River-3 | 38.2 | Trial pit 3-leach-Sea-3 | 41.0 |
| Trial pit 3-leach-River-4 | 37.8 | Trial pit 3-leach-Sea-4 | 41.6 |
| Trial pit 3-leach-River-5 | 37.9 | Trial pit 3-leach-Sea-5 | 41.1 |
| Trial pit 3-leach-River-6 | 37.8 | Trial pit 3-leach-Sea-6 | 42.0 |
| Trial pit 3-leach-River-7 | 38.6 | Trial pit 3-leach-Sea-7 | 39.7 |
| Trial pit 3-leach-River-8 | 37.9 | Trial pit 3-leach-Sea-8 | 38.8 |
| Trial pit 3-leach-River-9 | 39.6 | Trial pit 3-leach-Sea-9 | 40.6 |
| Trial pit 3-leach-River-10 | 38.0 | Trial pit 3-leach-Sea-10 | 42.1 |
| Trial pit 3-leach-River-11 | 36.3 | Trial pit 3-leach-Sea-11 | 40.7 |
| Trial pit 3-leach-River-12 | 41.5 | Trial pit 3-leach-Sea-12 | 40.5 |

**Table A.27: Metal concentrations in the leachates (µg $L^{-1}$), not blank adjusted - Leigh Marshes**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 5-leach-River-1 | 4.959 | 0.128 | 0.414 | 0.165 | 1.991 | 3.317 | 0.731 | 6.165 | 0.138 | 47.380 |
| Trial pit 5-leach-River-2 | 3.867 | 0.250 | 0.485 | 0.307 | 2.255 | 6.016 | 0.593 | 8.654 | 0.366 | 96.650 |
| Trial pit 5-leach-River-3 | 11.240 | 0.234 | 0.445 | 0.464 | 1.999 | 5.269 | 0.272 | 6.482 | 0.270 | 109.700 |
| Trial pit 5-leach-River-4 | 7.158 | 0.356 | 0.527 | 0.734 | 2.711 | 4.191 | 0.048 | 7.016 | 0.412 | 124.400 |
| Trial pit 5-leach-River-5 | 8.666 | 0.245 | 0.442 | 0.476 | 3.529 | 3.400 | 0.021 | 7.004 | 0.397 | 136.000 |
| Trial pit 5-leach-River-6 | 5.900 | 0.194 | 0.428 | 0.265 | 1.740 | 4.627 |  | 4.208 | 0.603 | 56.360 |
| Trial pit 5-leach-River-7 | 13.550 | 0.233 | 0.469 | 0.364 | 1.697 | 5.590 | 0.107 | 6.134 | 0.331 | 105.600 |
| Trial pit 5-leach-River-8 | 9.776 | 0.253 | 0.442 | 0.382 | 2.431 | 4.917 | 0.065 | 6.011 | 0.594 | 107.500 |
| Trial pit 5-leach-River-9 | 19.060 | 0.253 | 0.477 | 0.556 | 2.248 | 5.219 | 0.185 | 7.128 | 0.934 | 129.000 |
| Trial pit 5-leach-River-10 | 2.870 | 0.167 | 0.191 | 0.220 | 0.894 | 2.214 | 0.168 | 2.544 | 0.184 | 45.090 |
| Trial pit 5-leach-River-11 | 32.770 | 0.290 | 0.464 | 0.478 | 3.978 | 5.269 | 0.214 | 7.030 | 3.883 | 151.400 |
| Trial pit 5-leach-River-12 | 8.902 | 0.244 | 0.492 | 0.363 | 2.649 | 5.085 | 0.066 | 7.503 | 0.433 | 144.200 |
| Trial pit 5-leach-River-Blank 1 | 1.843 | 0.039 | 0.032 | 0.082 | 0.304 | 1.337 | 0.695 | 0.795 | 0.221 | 18.920 |
| Trial pit 5-leach-River-Blank 3 | 0.767 | 0.015 | 0.014 | 0.018 | 0.328 | 0.979 | 1.133 | 0.180 | 0.064 | 4.604 |
| Trial pit 5-leach-Sea-1 | 28.220 | 7.565 | 1.550 | 1.535 | 22.860 | 21.940 | 11.530 | 34.300 | 101.500 | 614.100 |
| Trial pit 5-leach-Sea-2 | 14.830 | 7.057 | 1.090 | 1.742 | 12.110 | 22.910 | 5.789 | 30.050 | 24.010 | 538.100 |
| Trial pit 5-leach-Sea-3 | 10.200 | 5.137 | 0.727 | 1.510 | 6.033 | 15.400 | 4.287 | 20.680 | 17.680 | 442.800 |
| Trial pit 5-leach-Sea-4 | 23.590 | 7.277 | 1.043 | 1.551 | 13.620 | 20.970 | 5.378 | 29.680 | 27.130 | 587.200 |
| Trial pit 5-leach-Sea-5 | 11.670 | 7.167 | 0.991 | 2.046 | 13.190 | 20.730 | 5.951 | 31.270 | 8.647 | 541.100 |
| Trial pit 5-leach-Sea-6 | 31.960 | 7.079 | 1.126 | 1.486 | 14.890 | 20.860 | 5.926 | 31.440 | 47.270 | 582.400 |
| Trial pit 5-leach-Sea-7 | 31.960 | 7.186 | 1.100 | 1.983 | 13.440 | 19.760 | 6.348 | 30.440 | 22.860 | 546.300 |
| Trial pit 5-leach-Sea-8 | 16.560 | 7.116 | 1.209 | 1.833 | 11.710 | 20.080 | 7.130 | 30.390 | 42.690 | 614.900 |
| Trial pit 5-leach-Sea-9 | 21.530 | 6.850 | 0.952 | 1.834 | 9.836 | 19.560 | 5.644 | 28.670 | 18.250 | 525.400 |
| Trial pit 5-leach-Sea-10 | 30.540 | 7.300 | 1.057 | 1.418 | 12.190 | 19.790 | 4.581 | 27.630 | 23.680 | 538.300 |
| Trial pit 5-leach-Sea-11 | 15.270 | 6.295 | 1.005 | 1.581 | 11.590 | 18.730 | 4.759 | 26.950 | 12.730 | 422.700 |
| Trial pit 5-leach-Sea-12 | 30.000 | 7.367 | 1.071 | 1.924 | 12.560 | 19.040 | 5.105 | 30.820 | 24.860 | 496.900 |
| Trial pit 5-leach-Sea-Blank 1 | 8.535 | 0.096 | 0.104 | 1.303 | 1.375 | 13.290 | 28.500 | 2.012 | 0.245 | 39.780 |
| Trial pit 5-leach-Sea-Blank 3 | 14.710 | 0.096 | 0.103 | 1.111 | 1.379 | 11.800 | 29.010 | 2.575 | 0.673 | 57.830 |

**Table A.28: Metal concentrations in the leachates (µg $L^{-1}$), not blank adjusted - Hadleigh Marsh**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 3-leach-River-1 | 16.460 | 0.224 | 0.168 | 0.572 | 4.087 | 4.443 | 0.208 | 3.957 | 0.209 | 63.430 |
| Trial pit 3-leach-River-2 | 15.570 | 0.174 | 0.145 | 0.695 | 4.366 | 4.804 | 0.150 | 3.967 | 0.176 | 58.930 |
| Trial pit 3-leach-River-3 | 12.920 | 0.153 | 0.142 | 0.532 | 4.844 | 6.160 | 0.159 | 3.853 | 0.129 | 65.470 |
| Trial pit 3-leach-River-4 | 32.510 | 0.227 | 0.184 | 0.585 | 4.545 | 5.871 | 0.129 | 4.303 | 0.439 | 76.640 |
| Trial pit 3-leach-River-5 | 10.540 | 0.262 | 0.192 | 0.689 | 4.264 | 5.220 | 0.139 | 3.987 | 0.149 | 63.710 |
| Trial pit 3-leach-River-6 | 27.820 | 0.261 | 0.151 | 0.624 | 5.480 | 5.408 | 0.200 | 5.087 | 0.341 | 80.110 |
| Trial pit 3-leach-River-7 | 12.480 | 0.146 | 0.150 | 0.732 | 4.835 | 6.596 | 0.202 | 4.048 | 0.160 | 44.470 |
| Trial pit 3-leach-River-8 | 32.530 | 0.181 | 0.137 | 0.762 | 4.641 | 6.615 | 0.197 | 3.545 | 0.140 | 72.490 |
| Trial pit 3-leach-River-9 | 28.550 | 0.256 | 0.139 | 0.615 | 5.260 | 4.766 | 0.150 | 4.214 | 0.404 | 83.160 |
| Trial pit 3-leach-River-10 | 28.070 | 0.174 | 0.113 | 0.518 | 4.800 | 4.770 | 0.102 | 3.504 | 0.331 | 61.100 |
| Trial pit 3-leach-River-11 | 29.400 | 0.262 | 0.133 | 0.657 | 4.402 | 5.607 | 0.135 | 4.733 | 0.310 | 73.250 |
| Trial pit 3-leach-River-12 | 14.400 | 0.222 | 0.165 | 1.214 | 4.709 | 11.310 | 0.293 | 4.147 | 0.249 | 44.190 |
| Trial pit 3-leach-River-Blank 2 | 5.849 | 0.055 | 0.035 | 0.083 | 0.542 | 3.398 | 0.792 | 1.027 | 0.123 | 52.870 |
| Trial pit 3-leach-River-Blank 4 | 2.324 | 0.060 | 0.050 | 0.078 | 3.113 | 1.829 | 0.433 | 0.346 | 0.079 | 36.500 |
| Trial pit 3-leach-Sea-1 | 25.580 | 9.193 | 0.980 | 5.375 | 24.600 | 29.140 | 9.089 | 21.320 | 4.985 | 527.100 |
| Trial pit 3-leach-Sea-2 | 12.290 | 7.990 | 0.532 | 4.483 | 16.510 | 27.760 | 3.051 | 20.030 | 1.210 | 377.900 |
| Trial pit 3-leach-Sea-3 | 11.680 | 8.773 | 0.490 | 3.318 | 13.090 | 29.120 | 3.715 | 22.780 | 4.772 | 523.400 |
| Trial pit 3-leach-Sea-4 | 8.859 | 9.478 | 0.329 | 3.032 | 11.930 | 26.590 | 2.906 | 18.820 | 1.297 | 492.400 |
| Trial pit 3-leach-Sea-5 | 31.220 | 10.560 | 0.436 | 2.913 | 19.820 | 28.360 | 5.010 | 24.550 | 5.049 | 694.600 |
| Trial pit 3-leach-Sea-6 | 15.360 | 9.748 | 0.391 | 2.269 | 13.480 | 25.920 | 3.937 | 22.700 | 2.021 | 499.500 |
| Trial pit 3-leach-Sea-7 | 6.364 | 1.758 | 0.118 | 0.798 | 3.179 | 11.360 | 0.888 | 4.213 | 0.834 | 125.400 |
| Trial pit 3-leach-Sea-8 | 28.920 | 16.710 | 0.688 | 4.143 | 26.210 | 41.310 | 8.470 | 38.040 | 4.381 | 809.200 |
| Trial pit 3-leach-Sea-9 | 23.980 | 9.372 | 0.288 | 2.108 | 15.870 | 23.110 | 3.267 | 19.240 | 3.069 | 365.000 |
| Trial pit 3-leach-Sea-10 | 15.070 | 9.686 | 0.429 | 2.397 | 15.580 | 23.600 | 3.324 | 20.390 | 1.317 | 460.400 |
| Trial pit 3-leach-Sea-11 | 24.230 | 8.187 | 0.309 | 2.252 | 11.680 | 22.350 | 2.561 | 16.880 | 2.014 | 382.800 |
| Trial pit 3-leach-Sea-12 | 5.379 | 0.046 | 0.019 | 0.387 | 1.081 | 8.513 | 0.285 | 0.536 | 0.155 | 34.510 |
| Trial pit 3-leach-Sea-Blank 2 | 13.380 | 0.060 | 0.107 | 1.591 | 1.278 | 18.350 | 27.120 | 2.482 | 0.456 | 41.370 |
| Trial pit 3-leach-Sea-Blank 4 | 12.460 | 0.314 | 0.331 | 1.779 | 1.729 | 20.020 | 26.230 | 2.973 | 1.040 | 45.840 |

**Table A.29: Concentrations of metals (mg kg$^{-1}$) in dried post-leaching matrix material - Leigh Marshes deionised water leachates**

| Leachate name | Post-leaching matrix material sub-sample | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 5-leach-River-1 | TP5-LRD-1-River-A | 10472 | 1.92 | 29.6 | 32.2 | 1394 | 79377 | 742 | 114.4 | 529 | 1117 |
| | TP5-LRD-1-River-B | 10502 | 2.49 | 29.4 | 13.2 | 3070 | 84020 | 618 | 114.2 | 10256 | 1291 |
| | TP5-LRD-1-River-C | 11084 | 2.90 | 27.5 | 20.8 | 423 | 82103 | 692 | 101.9 | 1598 | 1434 |
| Trial pit 5-leach-River-2 | TP5-LRD-2-River-A | 11856 | 1.97 | 22.3 | 13.6 | 401 | 65817 | 678 | 94.1 | 668 | 1031 |
| | TP5-LRD-2-River-B | 10846 | 1.60 | 26.4 | 23.2 | 1348 | 54613 | 526 | 92.2 | 1189 | 1181 |
| | TP5-LRD-2-River-C | 14603 | 2.68 | 75.1 | 96.5 | 774 | 84992 | 1146 | 241.2 | 2655 | 1700 |
| Trial pit 5-leach-River-3 | TP5-LRD-3-River-A | 10412 | 2.71 | 30.6 | 28.8 | 474 | 83733 | 833 | 118.5 | 2317 | 1172 |
| | TP5-LRD-3-River-B | 11431 | 2.01 | 41.1 | 37.8 | 409 | 81728 | 815 | 145.7 | 882 | 1016 |
| | TP5-LRD-3-River-C | 11753 | 2.09 | 39.9 | 37.9 | 434 | 75508 | 815 | 136.2 | 1032 | 1179 |
| Trial pit 5-leach-River-4 | TP5-LRD-4-River-A | 12797 | 2.25 | 53.2 | 72.6 | 632 | 71059 | 848 | 174.6 | 1289 | 1782 |
| | TP5-LRD-4-River-B | 12377 | 2.75 | 44.0 | 40.2 | 468 | 75002 | 1050 | 158.8 | 899 | 3530 |
| | TP5-LRD-4-River-C | 13208 | 2.49 | 55.8 | 66.6 | 593 | 84216 | 1025 | 192.8 | 1282 | 1785 |
| Trial pit 5-leach-River-5 | TP5-LRD-5-River-A | 11662 | 2.90 | 44.0 | 46.4 | 488 | 63767 | 9660 | 156.5 | 1713 | 1632 |
| | TP5-LRD-5-River-B | 13511 | 2.89 | 62.9 | 68.1 | 692 | 80276 | 1127 | 202.5 | 1510 | 1445 |
| | TP5-LRD-5-River-C | 11050 | 3.05 | 32.4 | 36.9 | 374 | 62467 | 864 | 112.2 | 841 | 2259 |
| Trial pit 5-leach-River-6 | TP5-LRD-6-River-A | 10913 | 2.28 | 39.6 | 52.0 | 456 | 64793 | 730 | 132.0 | 1286 | 1033 |
| | TP5-LRD-6-River-B | 9715 | 2.39 | 33.7 | 26.9 | 335 | 73733 | 839 | 118.6 | 1022 | 1031 |
| | TP5-LRD-6-River-C | 11821 | 2.79 | 37.5 | 33.9 | 656 | 76289 | 1026 | 121.6 | 1047 | 1198 |
| Trial pit 5-leach-River-7 | TP5-LRD-7-River-A | 12024 | 2.89 | 46.7 | 47.0 | 469 | 81052 | 987 | 146.2 | 1071 | 1341 |
| | TP5-LRD-7-River-B | 10822 | 2.91 | 38.4 | 22.9 | 704 | 101501 | 1035 | 137.5 | 845 | 1165 |
| | TP5-LRD-7-River-C | 11869 | 2.55 | 39.6 | 39.8 | 467 | 84682 | 808 | 131.2 | 1026 | 1075 |
| Trial pit 5-leach-River-8 | TP5-LRD-8-River-A | 12345 | 2.59 | 50.9 | 53.8 | 633 | 68540 | 1007 | 160.9 | 1303 | 1284 |
| | TP5-LRD-8-River-B | 11776 | 2.64 | 44.3 | 39.0 | 534 | 73712 | 840 | 148.6 | 1015 | 1135 |
| | TP5-LRD-8-River-C | 12392 | 2.67 | 43.6 | 51.8 | 763 | 77777 | 841 | 144.8 | 1169 | 1309 |
| Trial pit 5-leach-River-9 | TP5-LRD-9-River-A | 12709 | 2.78 | 58.0 | 63.9 | 573 | 65429 | 990 | 197.2 | 1397 | 1644 |
| | TP5-LRD-9-River-B | 11196 | 2.74 | 46.7 | 42.0 | 460 | 89911 | 1271 | 155.2 | 955 | 1303 |
| | TP5-LRD-9-River-C | 11879 | 2.41 | 42.2 | 34.9 | 949 | 82997 | 841 | 142.8 | 1043 | 1140 |
| Trial pit 5-leach-River-10 | TP5-LRD-10-River-A | 13828 | 3.13 | 79.6 | 113.8 | 771 | 81831 | 1276 | 247.9 | 2396 | 2111 |
| | TP5-LRD-10-River-B | 13914 | 2.84 | 58.5 | 71.4 | 656 | 76895 | 1005 | 183.7 | 2016 | 1597 |
| | TP5-LRD-10-River-C | 23223 | 3.53 | 91.5 | 136.0 | 816 | 104204 | 1546 | 281.2 | 3033 | 2552 |
| Trial pit 5-leach-River-11 | TP5-LRD-11-River-A | 14767 | 3.12 | 86.7 | 130.4 | 836 | 83047 | 1280 | 269.4 | 2704 | 2002 |
| | TP5-LRD-11-River-B | 13915 | 2.94 | 69.2 | 98.0 | 781 | 78391 | 1068 | 214.8 | 2426 | 1613 |
| | TP5-LRD-11-River-C | 14929 | 2.95 | 70.3 | 93.0 | 780 | 80379 | 1074 | 218.9 | 2453 | 1622 |
| Trial pit 5-leach-River-12 | TP5-LRD-12-River-A | 14213 | 3.01 | 80.3 | 117.3 | 807 | 78678 | 1209 | 248.6 | 2040 | 1850 |
| | TP5-LRD-12-River-B sample spilt | | | | | | | | | | |
| | TP5-LRD-12-River-C | 14153 | 2.92 | 72.9 | 96.8 | 781 | 78595 | 1078 | 227.0 | 1876 | 1652 |

**Table A.30: Concentrations of metals (mg kg$^{-1}$) in dried post-leaching matrix material - Leigh Marshes artificial seawater leachates**

| Leachate name | Post-leaching matrix material sub-sample | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 5-leach-Sea-1 | TP5-LRD-1-Sea-A | 10268 | 2.27 | 31.0 | 20.0 | 7848 | 80632 | 774 | 117.6 | 1548 | 987 |
| | TP5-LRD-1-Sea-B | 10595 | 2.51 | 33.4 | 27.5 | 417 | 68363 | 852 | 107.1 | 1537 | 1163 |
| | TP5-LRD-1-Sea-C | 11952 | 2.28 | 42.2 | 40.4 | 455 | 78620 | 786 | 138.0 | 1640 | 1103 |
| Trial pit 5-leach-Sea-2 | TP5-LRD-2-Sea-A | 11594 | 2.38 | 36.1 | 27.4 | 372 | 72738 | 791 | 126.6 | 848 | 997 |
| | TP5-LRD-2-Sea-B | 12342 | 2.33 | 50.7 | 40.9 | 435 | 69140 | 950 | 155.4 | 1173 | 1251 |
| | TP5-LRD-2-Sea-C | 11078 | 1.94 | 26.8 | 15.8 | 693 | 70969 | 780 | 89.9 | 736 | 859 |
| Trial pit 5-leach-Sea-3 | TP5-LRD-3-Sea-A | 10208 | 2.35 | 25.6 | 13.3 | 261 | 82178 | 437 | 88.6 | 792 | 872 |
| | TP5-LRD-3-Sea-B | 11705 | 3.67 | 50.5 | 49.3 | 501 | 74661 | 826 | 174.8 | 1848 | 2426 |
| | TP5-LRD-3-Sea-C | 11037 | 2.45 | 33.6 | 28.0 | 566 | 66838 | 723 | 108.6 | 1517 | 1158 |
| Trial pit 5-leach-Sea-4 | TP5-LRD-4-Sea-A | 11559 | 3.85 | 40.7 | 32.8 | 497 | 82846 | 830 | 144.9 | 943 | 1073 |
| | TP5-LRD-4-Sea-B | 12475 | 4.46 | 41.9 | 28.6 | 518 | 88393 | 842 | 147.4 | 1088 | 1037 |
| | TP5-LRD-4-Sea-C | 11238 | 2.50 | 36.9 | 16.0 | 376 | 83243 | 958 | 128.7 | 1051 | 944 |
| Trial pit 5-leach-Sea-5 | TP5-LRD-5-Sea-A | 11018 | 2.03 | 34.4 | 32.5 | 1133 | 62729 | 704 | 120.4 | 953 | 927 |
| | TP5-LRD-5-Sea-B | 10762 | 2.12 | 30.9 | 24.5 | 358 | 70386 | 802 | 104.9 | 698 | 967 |
| | TP5-LRD-5-Sea-C | 11691 | 2.48 | 40.2 | 34.5 | 508 | 79451 | 870 | 171.2 | 842 | 1068 |
| Trial pit 5-leach-Sea-6 | TP5-LRD-6-Sea-A | 13233 | 2.39 | 41.9 | 44.0 | 516 | 67938 | 805 | 138.1 | 1985 | 1182 |
| | TP5-LRD-6-Sea-B | 12913 | 2.54 | 42.2 | 41.5 | 449 | 72697 | 987 | 134.9 | 1209 | 1159 |
| | TP5-LRD-6-Sea-C | 11443 | 2.68 | 31.4 | 26.6 | 1637 | 69775 | 774 | 107.6 | 4407 | 1606 |
| Trial pit 5-leach-Sea-7 | TP5-LRD-7-Sea-A | 11972 | 2.66 | 35.9 | 30.8 | 436 | 73178 | 671 | 117.5 | 929 | 1035 |
| | TP5-LRD-7-Sea-B | 14193 | 3.22 | 57.5 | 65.6 | 581 | 72013 | 942 | 182.9 | 1298 | 1423 |
| | TP5-LRD-7-Sea-C | 13093 | 2.48 | 53.5 | 64.9 | 645 | 71904 | 906 | 182.9 | 1068 | 1259 |
| Trial pit 5-leach-Sea-8 | TP5-LRD-8-Sea-A | 12879 | 2.69 | 46.8 | 45.1 | 421 | 76670 | 1753 | 152.1 | 1339 | 1343 |
| | TP5-LRD-8-Sea-B | 10996 | 2.21 | 29.8 | 66.2 | 402 | 63022 | 567 | 106.9 | 934 | 995 |
| | TP5-LRD-8-Sea-C | 12002 | 3.22 | 38.3 | 36.2 | 400 | 78719 | 766 | 128.6 | 2700 | 1632 |
| Trial pit 5-leach-Sea-9 | TP5-LRD-9-Sea-A | 13637 | 2.41 | 53.9 | 63.0 | 466 | 67464 | 880 | 166.6 | 1156 | 1515 |
| | TP5-LRD-9-Sea-B | 13157 | 2.57 | 46.3 | 43.5 | 455 | 80235 | 827 | 151.4 | 1153 | 1199 |
| | TP5-LRD-9-Sea-C | 12908 | 2.97 | 65.1 | 67.6 | 586 | 80677 | 1097 | 212.2 | 1296 | 1482 |
| Trial pit 5-leach-Sea-10 | TP5-LRD-10-Sea-A | 14346 | 2.73 | 52.5 | 63.6 | 540 | 71669 | 866 | 161.7 | 1426 | 1281 |
| | TP5-LRD-10-Sea-B | 15073 | 2.95 | 75.0 | 105.7 | 624 | 75123 | 1094 | 225.6 | 1799 | 1709 |
| | TP5-LRD-10-Sea-C | 14846 | 2.87 | 72.2 | 100.7 | 635 | 74505 | 1096 | 217.5 | 1798 | 1691 |
| Trial pit 5-leach-Sea-11 | TP5-LRD-11-Sea-A | 13534 | 2.59 | 51.1 | 59.7 | 615 | 67199 | 871 | 157.5 | 1496 | 1288 |
| | TP5-LRD-11-Sea-B | 14026 | 2.59 | 51.6 | 66.0 | 613 | 69639 | 860 | 163.8 | 1482 | 1260 |
| | TP5-LRD-11-Sea-C | 14848 | 2.89 | 69.2 | 101.5 | 672 | 71861 | 1043 | 214.6 | 1783 | 1659 |
| Trial pit 5-leach-Sea-12 | TP5-LRD-12-Sea-A | 14455 | 2.97 | 63.8 | 85.8 | 660 | 74443 | 1048 | 195.0 | 1699 | 1536 |
| | TP5-LRD-12-Sea-B | 14515 | 2.92 | 54.2 | 63.5 | 640 | 73403 | 931 | 166.2 | 1514 | 1325 |
| | TP5-LRD-12-Sea-C | 14991 | 3.02 | 63.0 | 79.2 | 655 | 75803 | 1012 | 190.4 | 1660 | 1502 |

**Table A.31: Concentrations of metals (mg kg$^{-1}$) in dried post-leaching matrix material - Hadleigh Marsh deionised water leachates**

| Leachate name | Post-leaching matrix material sub-sample | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 3-leach-River-1 | TP3-LRD-1-River-A | 11718 | 1.59 | 16.9 | 63.2 | 112 | 38287 | 444 | 53.4 | 522 | 1329 |
| | TP3-LRD-1-River-B | 10940 | 1.82 | 17.4 | 60.9 | 128 | 36353 | 477 | 52.5 | 574 | 1389 |
| | TP3-LRD-1-River-C | 12420 | 1.82 | 19.4 | 72.6 | 141 | 41106 | 494 | 60.7 | 608 | 1541 |
| Trial pit 3-leach-River-2 | TP3-LRD-2-River-A | 10397 | 1.78 | 17.3 | 63.5 | 121 | 36922 | 487 | 50.0 | 639 | 1456 |
| | TP3-LRD-2-River-B | 11110 | 1.84 | 19.9 | 74.1 | 138 | 41526 | 540 | 60.1 | 753 | 1733 |
| | TP3-LRD-2-River-C | 11774 | 2.00 | 21.1 | 80.4 | 164 | 40902 | 578 | 64.3 | 787 | 1748 |
| Trial pit 3-leach-River-3 | TP3-LRD-3-River-A | 10417 | 1.86 | 15.8 | 52.5 | 173 | 34908 | 447 | 48.6 | 606 | 1528 |
| | TP3-LRD-3-River-B | 12323 | 2.42 | 24.0 | 87.1 | 261 | 45330 | 659 | 76.0 | 902 | 2255 |
| | TP3-LRD-3-River-C | 11935 | 2.09 | 19.3 | 70.2 | 179 | 40354 | 508 | 58.6 | 707 | 1831 |
| Trial pit 3-leach-River-4 | TP3-LRD-4-River-A | 12068 | 1.69 | 17.2 | 64.4 | 100 | 38464 | 446 | 54.4 | 590 | 1463 |
| | TP3-LRD-4-River-B | 10126 | 1.73 | 15.3 | 48.4 | 88 | 31901 | 413 | 46.3 | 561 | 1340 |
| | TP3-LRD-4-River-C | 11550 | 1.71 | 18.4 | 68.0 | 112 | 40014 | 473 | 59.2 | 627 | 1559 |
| Trial pit 3-leach-River-5 | TP3-LRD-5-River-A | 11784 | 1.84 | 18.2 | 65.9 | 123 | 39801 | 497 | 57.8 | 612 | 1649 |
| | TP3-LRD-5-River-B | 11724 | 1.85 | 17.5 | 65.9 | 119 | 39855 | 465 | 56.8 | 586 | 1624 |
| | TP3-LRD-5-River-C | 11350 | 1.78 | 18.3 | 65.5 | 134 | 39978 | 516 | 62.8 | 665 | 1833 |
| Trial pit 3-leach-River-6 | TP3-LRD-6-River-A | 10749 | 1.75 | 16.3 | 54.8 | 93 | 35537 | 455 | 52.2 | 539 | 1470 |
| | TP3-LRD-6-River-B | 11348 | 1.92 | 17.1 | 64.1 | 115 | 38460 | 453 | 55.0 | 581 | 1544 |
| | TP3-LRD-6-River-C | 11584 | 1.82 | 18.4 | 74.2 | 139 | 40033 | 494 | 60.2 | 613 | 1610 |
| Trial pit 3-leach-River-7 | TP3-LRD-7-River-A | 9959 | 1.41 | 13.3 | 44.9 | 74 | 30361 | 358 | 40.2 | 520 | 1169 |
| | TP3-LRD-7-River-B | 11949 | 1.82 | 20.1 | 71.7 | 116 | 40868 | 513 | 58.6 | 738 | 1616 |
| | TP3-LRD-7-River-C | 12497 | 1.86 | 23.4 | 83.0 | 121 | 46490 | 592 | 65.9 | 821 | 1788 |
| Trial pit 3-leach-River-8 | TP3-LRD-8-River-A | 11763 | 1.45 | 15.5 | 54.7 | 91 | 35407 | 415 | 44.0 | 510 | 1402 |
| | TP3-LRD-8-River-B | 11259 | 1.71 | 17.9 | 58.7 | 113 | 37850 | 488 | 52.7 | 608 | 1655 |
| | TP3-LRD-8-River-C | 10668 | 1.68 | 15.9 | 55.4 | 118 | 33545 | 451 | 48.3 | 553 | 1497 |
| Trial pit 3-leach-River-9 | TP3-LRD-9-River-A | 12053 | 1.97 | 18.7 | 69.3 | 131 | 40473 | 470 | 58.3 | 570 | 1614 |
| | TP3-LRD-9-River-B | 12640 | 1.86 | 19.9 | 78.1 | 145 | 43187 | 487 | 62.5 | 603 | 1753 |
| | TP3-LRD-9-River-C | 11250 | 2.19 | 16.4 | 58.0 | 133 | 35485 | 455 | 52.4 | 541 | 1490 |
| Trial pit 3-leach-River-10 | TP3-LRD-10-River-A | 11935 | 1.61 | 18.0 | 64.9 | 112 | 37090 | 472 | 53.7 | 448 | 1220 |
| | TP3-LRD-10-River-B | 11965 | 1.43 | 16.3 | 56.5 | 91 | 34387 | 429 | 45.7 | 398 | 1054 |
| | TP3-LRD-10-River-C | 13549 | 1.22 | 16.8 | 70.4 | 73 | 39503 | 379 | 49.1 | 344 | 1045 |
| Trial pit 3-leach-River-11 | TP3-LRD-11-River-A | 12957 | 1.71 | 19.4 | 72.9 | 109 | 42789 | 487 | 64.2 | 590 | 1589 |
| | TP3-LRD-11-River-B | 10841 | 1.78 | 15.4 | 52.5 | 103 | 33050 | 438 | 50.0 | 542 | 1302 |
| | TP3-LRD-11-River-C | 13350 | 1.65 | 19.8 | 80.4 | 102 | 45459 | 477 | 66.7 | 579 | 1619 |
| Trial pit 3-leach-River-12 | TP3-LRD-12-River-A | 11058 | 1.54 | 18.6 | 63.8 | 121 | 37901 | 471 | 54.3 | 713 | 1588 |
| | TP3-LRD-12-River-B | 11736 | 1.66 | 19.5 | 70.0 | 135 | 38454 | 516 | 55.4 | 741 | 1651 |
| | TP3-LRD-12-River-C | 13015 | 2.02 | 25.6 | 100.2 | 173 | 49093 | 644 | 76.2 | 975 | 2199 |

**Table A.32: Concentrations of metals (mg kg$^{-1}$) in dried post-leaching matrix material - Hadleigh Marsh artificial seawater leachates**

| Leachate name | Post-leaching matrix material sub-sample | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 3-leach-Sea-1 | TP3-LRD-1-Sea-A | 9802 | 1.67 | 15.3 | 46.8 | 146 | 31645 | 428 | 46.2 | 613 | 1387 |
| | TP3-LRD-1-Sea-B | 10831 | 1.84 | 18.2 | 57.6 | 184 | 35484 | 508 | 55.7 | 710 | 1641 |
| | TP3-LRD-1-Sea-C | 13265 | 1.90 | 24.1 | 85.5 | 234 | 45389 | 617 | 73.2 | 894 | 2042 |
| Trial pit 3-leach-Sea-2 | TP3-LRD-2-Sea-A | 11268 | 1.64 | 15.5 | 53.6 | 109 | 35043 | 459 | 45.2 | 493 | 1183 |
| | TP3-LRD-2-Sea-B | 11536 | 1.63 | 17.1 | 58.6 | 124 | 37130 | 487 | 50.2 | 530 | 1271 |
| | TP3-LRD-2-Sea-C | 12011 | 1.78 | 17.8 | 63.2 | 130 | 38327 | 505 | 52.3 | 564 | 1322 |
| Trial pit 3-leach-Sea-3 | TP3-LRD-3-Sea-A | 9693 | 1.58 | 13.3 | 43.3 | 84 | 29541 | 366 | 37.9 | 554 | 1167 |
| | TP3-LRD-3-Sea-B | 10644 | 1.76 | 16.3 | 56.7 | 128 | 32159 | 462 | 49.1 | 710 | 1404 |
| | TP3-LRD-3-Sea-C | 13340 | 1.91 | 23.7 | 84.2 | 166 | 44504 | 627 | 70.0 | 966 | 1922 |
| Trial pit 3-leach-Sea-4 | TP3-LRD-4-Sea-A | 9536 | 1.51 | 13.8 | 43.5 | 86 | 30559 | 415 | 40.9 | 468 | 1135 |
| | TP3-LRD-4-Sea-B | 9428 | 1.58 | 13.3 | 40.1 | 82 | 29443 | 406 | 36.8 | 469 | 1098 |
| | TP3-LRD-4-Sea-C | 14278 | 1.57 | 19.9 | 83.1 | 114 | 43553 | 499 | 60.3 | 595 | 1532 |
| Trial pit 3-leach-Sea-5 | TP3-LRD-5-Sea-A | 9836 | 1.53 | 13.4 | 42.9 | 79 | 31111 | 381 | 37.3 | 479 | 1262 |
| | TP3-LRD-5-Sea-B | 11750 | 1.70 | 17.2 | 58.7 | 113 | 35717 | 472 | 48.1 | 604 | 1519 |
| | TP3-LRD-5-Sea-C | 13126 | 1.98 | 20.9 | 84.8 | 167 | 42285 | 571 | 64.6 | 765 | 1892 |
| Trial pit 3-leach-Sea-6 | TP3-LRD-6-Sea-A | 11478 | 1.83 | 17.6 | 59.6 | 134 | 35167 | 489 | 51.2 | 613 | 1459 |
| | TP3-LRD-6-Sea-B | 13139 | 1.93 | 21.4 | 77.1 | 139 | 43525 | 565 | 63.3 | 715 | 1742 |
| | TP3-LRD-6-Sea-C | 10288 | 1.88 | 16.9 | 53.8 | 117 | 35271 | 473 | 48.4 | 595 | 1441 |
| Trial pit 3-leach-Sea-7 | TP3-LRD-7-Sea-A | 12556 | 1.82 | 17.7 | 62.2 | 104 | 36554 | 471 | 50.0 | 594 | 1598 |
| | TP3-LRD-7-Sea-B | 13094 | 1.95 | 18.9 | 69.2 | 116 | 38996 | 483 | 55.7 | 644 | 1739 |
| | TP3-LRD-7-Sea-C | 12583 | 1.93 | 20.0 | 71.8 | 146 | 38640 | 534 | 60.3 | 689 | 1821 |
| Trial pit 3-leach-Sea-8 | TP3-LRD-8-Sea-A | 9689 | 1.40 | 13.0 | 39.6 | 70 | 30044 | 361 | 36.9 | 427 | 1023 |
| | TP3-LRD-8-Sea-B | 10880 | 2.20 | 16.5 | 50.3 | 92 | 34110 | 444 | 46.6 | 560 | 1494 |
| | TP3-LRD-8-Sea-C | 12883 | 1.68 | 18.7 | 66.7 | 111 | 38942 | 490 | 56.6 | 631 | 1499 |
| Trial pit 3-leach-Sea-9 | TP3-LRD-9-Sea-A | 10902 | 1.65 | 13.7 | 45.5 | 95 | 31111 | 396 | 42.5 | 394 | 1003 |
| | TP3-LRD-9-Sea-B | 12890 | 1.65 | 16.6 | 59.8 | 107 | 37758 | 429 | 52.5 | 441 | 1201 |
| | TP3-LRD-9-Sea-C | 12178 | 1.66 | 15.8 | 55.3 | 98 | 35453 | 428 | 49.8 | 413 | 1107 |
| Trial pit 3-leach-Sea-10 | TP3-LRD-10-Sea-A | 11511 | 1.77 | 15.3 | 49.9 | 123 | 32585 | 402 | 46.1 | 464 | 1283 |
| | TP3-LRD-10-Sea-B | 11328 | 1.70 | 15.6 | 48.4 | 129 | 32354 | 417 | 46.4 | 475 | 1309 |
| | TP3-LRD-10-Sea-C | 12879 | 1.87 | 20.5 | 76.1 | 176 | 42878 | 514 | 65.6 | 616 | 1715 |
| Trial pit 3-leach-Sea-11 | TP3-LRD-11-Sea-A | 13155 | 1.28 | 17.0 | 63.8 | 89 | 40051 | 424 | 50.8 | 467 | 1296 |
| | TP3-LRD-11-Sea-B | 10908 | 1.50 | 15.0 | 45.0 | 92 | 32139 | 427 | 42.0 | 471 | 1166 |
| | TP3-LRD-11-Sea-C | 12138 | 1.55 | 17.0 | 58.6 | 110 | 37601 | 456 | 51.2 | 517 | 1350 |
| Trial pit 3-leach-Sea-12 | TP3-LRD-12-Sea-A | 10286 | 1.51 | 14.3 | 46.0 | 110 | 31341 | 386 | 40.6 | 475 | 1344 |
| | TP3-LRD-12-Sea-B | 12940 | 1.76 | 21.2 | 77.6 | 187 | 41842 | 538 | 63.3 | 688 | 1849 |
| | TP3-LRD-12-Sea-C | 12018 | 1.64 | 18.4 | 62.8 | 143 | 38406 | 471 | 52.7 | 587 | 1617 |

**Table A.33: Calculated initial (pre-leaching) concentrations of metals in matrix material from Leigh Marshes landfill (mg $kg^{-1}$)**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 5-leach-River-1 | 10686 | 2.44 | 28.9 | 22.1 | 1629 | 81833 | 684 | 110.3 | 4128 | 1281 |
| Trial pit 5-leach-River-2 | 12435 | 2.09 | 41.3 | 44.5 | 841 | 68474 | 783 | 142.6 | 1504 | 1305 |
| Trial pit 5-leach-River-3 | 11199 | 2.28 | 37.3 | 34.9 | 439 | 80323 | 821 | 133.5 | 1410 | 1123 |
| Trial pit 5-leach-River-4 | 12794 | 2.51 | 51.1 | 59.9 | 564 | 76759 | 975 | 175.5 | 1157 | 2367 |
| Trial pit 5-leach-River-5 | 12075 | 2.96 | 46.5 | 50.5 | 518 | 68836 | 3884 | 157.1 | 1355 | 1780 |
| Trial pit 5-leach-River-6 | 10816 | 2.50 | 37.0 | 37.7 | 482 | 71605 | 865 | 124.1 | 1119 | 1088 |
| Trial pit 5-leach-River-7 | 11572 | 2.79 | 41.6 | 36.6 | 547 | 89078 | 944 | 138.3 | 981 | 1194 |
| Trial pit 5-leach-River-8 | 12171 | 2.64 | 46.3 | 48.3 | 643 | 73343 | 896 | 151.5 | 1163 | 1243 |
| Trial pit 5-leach-River-9 | 11928 | 2.65 | 49.0 | 47.0 | 661 | 79446 | 1034 | 165.1 | 1132 | 1364 |
| Trial pit 5-leach-River-10 | 13871 | 2.99 | 69.1 | 92.6 | 713 | 79363 | 1141 | 215.8 | 2206 | 1854 |
| Trial pit 5-leach-River-11 | 14537 | 3.01 | 75.4 | 107.2 | 799 | 80606 | 1141 | 234.4 | 2528 | 1747 |
| Trial pit 5-leach-River-12 | 12240 | 2.65 | 55.7 | 68.7 | 4327 | 79655 | 992 | 183.4 | 1795 | 1419 |
| Trial pit 5-leach-Sea-1 | 11381 | 2.47 | 37.3 | 31.9 | 415 | 73241 | 810 | 124.6 | 1343 | 1093 |
| Trial pit 5-leach-Sea-2 | 11210 | 2.28 | 34.3 | 23.4 | 463 | 74096 | 722 | 112.0 | 901 | 998 |
| Trial pit 5-leach-Sea-3 | 11434 | 3.38 | 41.6 | 36.8 | 521 | 74782 | 793 | 143.4 | 1437 | 1556 |
| Trial pit 5-leach-Sea-4 | 11577 | 3.07 | 37.7 | 25.8 | 676 | 78122 | 835 | 132.9 | 1031 | 974 |
| Trial pit 5-leach-Sea-5 | 11896 | 2.40 | 37.7 | 34.5 | 461 | 72592 | 826 | 138.8 | 1176 | 1077 |
| Trial pit 5-leach-Sea-6 | 12110 | 2.70 | 36.5 | 33.1 | 841 | 71884 | 811 | 120.8 | 2182 | 1272 |
| Trial pit 5-leach-Sea-7 | 13389 | 2.87 | 52.6 | 58.7 | 549 | 73529 | 1201 | 173.3 | 1236 | 1346 |
| Trial pit 5-leach-Sea-8 | 12212 | 2.69 | 40.7 | 55.2 | 423 | 69735 | 738 | 134.8 | 1598 | 1386 |
| Trial pit 5-leach-Sea-9 | 13470 | 2.82 | 54.6 | 58.4 | 527 | 77527 | 930 | 175.8 | 1292 | 1325 |
| Trial pit 5-leach-Sea-10 | 14485 | 2.87 | 66.1 | 88.8 | 625 | 72276 | 1021 | 200.9 | 1698 | 1567 |
| Trial pit 5-leach-Sea-11 | 14443 | 2.88 | 61.6 | 84.6 | 648 | 71981 | 984 | 191.8 | 1655 | 1489 |
| Trial pit 5-leach-Sea-12 | 13541 | 26.76 | 42.9 | 184.4 | 470 | 57837 | 910 | 206.0 | 1114 | 1273 |

**Table A.34: Calculated initial (pre-leaching) concentrations of metals in matrix material from Hadleigh Marsh landfill**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 3-leach-River-1 | 11693 | 1.74 | 17.9 | 65.5 | 127 | 38582 | 472 | 55.6 | 568 | 1420 |
| Trial pit 3-leach-River-2 | 11094 | 1.87 | 19.4 | 72.6 | 141 | 39783 | 535 | 58.2 | 726 | 1646 |
| Trial pit 3-leach-River-3 | 11558 | 2.13 | 19.7 | 70.0 | 205 | 40197 | 538 | 61.1 | 738 | 1872 |
| Trial pit 3-leach-River-4 | 11249 | 1.71 | 17.0 | 60.3 | 100 | 36793 | 444 | 53.3 | 593 | 1455 |
| Trial pit 3-leach-River-5 | 11619 | 1.83 | 18.0 | 65.8 | 125 | 39878 | 493 | 59.2 | 621 | 1703 |
| Trial pit 3-leach-River-6 | 11227 | 1.83 | 17.3 | 64.4 | 116 | 38010 | 467 | 55.9 | 578 | 1542 |
| Trial pit 3-leach-River-7 | 11469 | 1.70 | 18.9 | 66.6 | 104 | 39240 | 487 | 54.9 | 693 | 1525 |
| Trial pit 3-leach-River-8 | 11230 | 1.62 | 16.4 | 56.3 | 108 | 35601 | 452 | 48.4 | 557 | 1518 |
| Trial pit 3-leach-River-9 | 11981 | 2.01 | 18.4 | 68.5 | 136 | 39715 | 471 | 57.8 | 571 | 1620 |
| Trial pit 3-leach-River-10 | 12484 | 1.42 | 17.0 | 63.9 | 92 | 36993 | 427 | 49.6 | 397 | 1107 |
| Trial pit 3-leach-River-11 | 12383 | 1.71 | 18.2 | 68.6 | 105 | 40433 | 467 | 60.3 | 571 | 1504 |
| Trial pit 3-leach-River-12 | 11937 | 1.74 | 21.2 | 78.0 | 143 | 41816 | 544 | 62.0 | 810 | 1813 |
| Trial pit 3-leach-Sea-1 | 11299 | 1.89 | 19.2 | 63.4 | 188 | 37506 | 518 | 58.6 | 739 | 1694 |
| Trial pit 3-leach-Sea-2 | 11605 | 1.76 | 16.8 | 58.5 | 121 | 36834 | 484 | 49.4 | 530 | 1262 |
| Trial pit 3-leach-Sea-3 | 11226 | 1.83 | 17.8 | 61.5 | 126 | 35402 | 485 | 52.6 | 744 | 1502 |
| Trial pit 3-leach-Sea-4 | 11081 | 1.65 | 15.7 | 55.7 | 94 | 34519 | 440 | 46.2 | 511 | 1259 |
| Trial pit 3-leach-Sea-5 | 11571 | 1.83 | 17.2 | 62.2 | 120 | 36371 | 474 | 50.2 | 616 | 1563 |
| Trial pit 3-leach-Sea-6 | 11635 | 1.97 | 18.6 | 63.6 | 130 | 37988 | 509 | 54.6 | 642 | 1551 |
| Trial pit 3-leach-Sea-7 | 12745 | 1.92 | 18.9 | 67.8 | 122 | 38063 | 496 | 55.4 | 643 | 1720 |
| Trial pit 3-leach-Sea-8 | 11151 | 1.91 | 16.1 | 52.3 | 91 | 34365 | 432 | 47.1 | 540 | 1345 |
| Trial pit 3-leach-Sea-9 | 11990 | 1.74 | 15.4 | 53.6 | 100 | 34774 | 418 | 48.5 | 417 | 1107 |
| Trial pit 3-leach-Sea-10 | 11906 | 1.87 | 17.1 | 58.2 | 143 | 35939 | 444 | 52.9 | 519 | 1439 |
| Trial pit 3-leach-Sea-11 | 12067 | 1.53 | 16.3 | 55.9 | 97 | 36597 | 436 | 48.2 | 485 | 1274 |
| Trial pit 3-leach-Sea-12 | 11748 | 1.65 | 18.0 | 62.2 | 147 | 37196 | 465 | 52.3 | 584 | 1604 |

**Table A.35: Proportions of metals released from Leigh Marshes landfill matrix material in 24 hours as a percentage of the initial concentrations of the metals in the matrix material**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| **Deionised water leachant** | | | | | | | | | | |
| Trial pit 5-leach-River-1 | 0.0005 | 0.0540 | 0.0148 | 0.0077 | 0.0013 | 0.0000 | 0.0011 | 0.0575 | 0.0000 | 0.0380 |
| Trial pit 5-leach-River-2 | 0.0003 | 0.1213 | 0.0119 | 0.0070 | 0.0027 | 0.0001 | 0.0008 | 0.0614 | 0.0002 | 0.0749 |
| Trial pit 5-leach-River-3 | 0.0010 | 0.1032 | 0.0120 | 0.0133 | 0.0046 | 0.0001 | 0.0003 | 0.0486 | 0.0002 | 0.0978 |
| Trial pit 5-leach-River-4 | 0.0006 | 0.1421 | 0.0103 | 0.0122 | 0.0048 | 0.0001 | 0.0000 | 0.0398 | 0.0004 | 0.0524 |
| Trial pit 5-leach-River-5 | 0.0007 | 0.0849 | 0.0097 | 0.0096 | 0.0070 | 0.0001 | 0.0000 | 0.0456 | 0.0003 | 0.0782 |
| Trial pit 5-leach-River-6 | 0.0005 | 0.0782 | 0.0116 | 0.0071 | 0.0036 | 0.0001 | | 0.0340 | 0.0005 | 0.0520 |
| Trial pit 5-leach-River-7 | 0.0011 | 0.0814 | 0.0110 | 0.0097 | 0.0030 | 0.0001 | 0.0001 | 0.0431 | 0.0003 | 0.0859 |
| Trial pit 5-leach-River-8 | 0.0008 | 0.0963 | 0.0096 | 0.0079 | 0.0038 | 0.0001 | 0.0001 | 0.0398 | 0.0005 | 0.0867 |
| Trial pit 5-leach-River-9 | 0.0016 | 0.0941 | 0.0096 | 0.0116 | 0.0033 | 0.0001 | 0.0002 | 0.0425 | 0.0008 | 0.0931 |
| Trial pit 5-leach-River-10 | 0.0002 | 0.0554 | 0.0027 | 0.0024 | 0.0012 | 0.0000 | 0.0001 | 0.0117 | 0.0001 | 0.0241 |
| Trial pit 5-leach-River-11 | 0.0023 | 0.0975 | 0.0062 | 0.0045 | 0.0050 | 0.0001 | 0.0002 | 0.0303 | 0.0016 | 0.0875 |
| Trial pit 5-leach-River-12 | 0.0006 | 0.0834 | 0.0065 | 0.0034 | 0.0034 | 0.0001 | 0.0001 | 0.0320 | 0.0002 | 0.0834 |
| **Minimum** | 0.0002 | 0.0540 | 0.0027 | 0.0024 | 0.0012 | 0.0000 | 0.0000 | 0.0117 | 0.0000 | 0.0241 |
| **Maximum** | 0.0023 | 0.1421 | 0.0148 | 0.0133 | 0.0070 | 0.0001 | 0.0011 | 0.0614 | 0.0016 | 0.0978 |
| **Range** | 0.0021 | 0.0880 | 0.0120 | 0.0110 | 0.0057 | 0.0001 | 0.0011 | 0.0497 | 0.0015 | 0.0737 |
| **Median** | 0.0007 | 0.0895 | 0.0100 | 0.0078 | 0.0035 | 0.0001 | 0.0002 | 0.0412 | 0.0003 | 0.0808 |
| **Artificial seawater leachant** | | | | | | | | | | |
| Trial pit 5-leach-Sea-1 | 0.0026 | 3.1908 | 0.0445 | 0.0535 | 0.0080 | 0.0003 | 0.0146 | 0.2890 | 0.0657 | 0.5755 |
| Trial pit 5-leach-Sea-2 | 0.0013 | 3.1162 | 0.0289 | 0.0624 | 0.0243 | 0.0003 | 0.0069 | 0.2431 | 0.0262 | 0.5198 |
| Trial pit 5-leach-Sea-3 | 0.0009 | 1.7947 | 0.0199 | 0.0501 | 0.0137 | 0.0002 | 0.0065 | 0.1669 | 0.0128 | 0.2981 |
| Trial pit 5-leach-Sea-4 | 0.0020 | 2.0062 | 0.0265 | 0.0607 | 0.0297 | 0.0002 | 0.0062 | 0.2133 | 0.0267 | 0.5798 |
| Trial pit 5-leach-Sea-5 | 0.0011 | 3.1790 | 0.0284 | 0.0675 | 0.0199 | 0.0003 | 0.0076 | 0.2376 | 0.0105 | 0.5490 |
| Trial pit 5-leach-Sea-6 | 0.0026 | 2.7317 | 0.0293 | 0.0399 | 0.0172 | 0.0003 | 0.0069 | 0.2479 | 0.0187 | 0.4420 |
| Trial pit 5-leach-Sea-7 | 0.0024 | 2.4889 | 0.0221 | 0.0363 | 0.0239 | 0.0003 | 0.0075 | 0.1859 | 0.0205 | 0.4330 |
| Trial pit 5-leach-Sea-8 | 0.0014 | 2.6108 | 0.0321 | 0.0379 | 0.0292 | 0.0003 | 0.0070 | 0.2386 | 0.0262 | 0.4704 |
| Trial pit 5-leach-Sea-9 | 0.0016 | 2.5528 | 0.0174 | 0.0319 | 0.0198 | 0.0003 | 0.0061 | 0.1635 | 0.0153 | 0.3778 |
| Trial pit 5-leach-Sea-10 | 0.0021 | 2.5045 | 0.0158 | 0.0157 | 0.0203 | 0.0003 | 0.0045 | 0.1367 | 0.0141 | 0.3435 |
| Trial pit 5-leach-Sea-11 | 0.0011 | 2.2704 | 0.0173 | 0.0206 | 0.0181 | 0.0003 | 0.0051 | 0.1490 | 0.0079 | 0.2972 |
| Trial pit 5-leach-Sea-12 | 0.0021 | 2.4582 | 0.0180 | 0.0256 | 0.0195 | 0.0003 | 0.0052 | 0.1695 | 0.0155 | 0.3451 |
| **Minimum** | 0.0009 | 1.7947 | 0.0159 | 0.0157 | 0.0080 | 0.0002 | 0.0045 | 0.1367 | 0.0079 | 0.2972 |
| **Maximum** | 0.0026 | 3.1908 | 0.0445 | 0.0675 | 0.0297 | 0.0003 | 0.0146 | 0.2890 | 0.0658 | 0.5798 |
| **Range** | 0.0017 | 1.3961 | 0.0287 | 0.0518 | 0.0217 | 0.0001 | 0.0102 | 0.1523 | 0.0578 | 0.2826 |
| **Median** | 0.0018 | 2.5287 | 0.0243 | 0.0389 | 0.0198 | 0.0003 | 0.0067 | 0.1996 | 0.0171 | 0.4375 |

**Table A.36: Proportions of metals released from Hadleigh Marsh landfill matrix material in 24 hours as a percentage of the initial concentrations of the metals in the matrix material**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| **Deionised water leachant** | | | | | | | | | | |
| Trial pit 3-leach-River-1 | 0.0014 | 0.1278 | 0.0093 | 0.0087 | 0.0320 | 0.0001 | 0.0004 | 0.0709 | 0.0004 | 0.0444 |
| Trial pit 3-leach-River-2 | 0.0014 | 0.0943 | 0.0076 | 0.0097 | 0.0314 | 0.0001 | 0.0003 | 0.0692 | 0.0002 | 0.0363 |
| Trial pit 3-leach-River-3 | 0.0011 | 0.0737 | 0.0074 | 0.0078 | 0.0243 | 0.0002 | 0.0003 | 0.0646 | 0.0002 | 0.0358 |
| Trial pit 3-leach-River-4 | 0.0029 | 0.1317 | 0.0108 | 0.0096 | 0.0453 | 0.0002 | 0.0003 | 0.0803 | 0.0007 | 0.0524 |
| Trial pit 3-leach-River-5 | 0.0009 | 0.1423 | 0.0106 | 0.0104 | 0.0337 | 0.0001 | 0.0003 | 0.0668 | 0.0002 | 0.0371 |
| Trial pit 3-leach-River-6 | 0.0025 | 0.1432 | 0.0088 | 0.0098 | 0.0477 | 0.0001 | 0.0004 | 0.0916 | 0.0006 | 0.0523 |
| Trial pit 3-leach-River-7 | 0.0011 | 0.0836 | 0.0077 | 0.0107 | 0.0453 | 0.0002 | 0.0004 | 0.0717 | 0.0002 | 0.0284 |
| Trial pit 3-leach-River-8 | 0.0028 | 0.1093 | 0.0081 | 0.0132 | 0.0421 | 0.0002 | 0.0004 | 0.0715 | 0.0002 | 0.0466 |
| Trial pit 3-leach-River-9 | 0.0024 | 0.1267 | 0.0075 | 0.0089 | 0.0384 | 0.0001 | 0.0003 | 0.0726 | 0.0007 | 0.0511 |
| Trial pit 3-leach-River-10 | 0.0023 | 0.1260 | 0.0068 | 0.0083 | 0.0536 | 0.0001 | 0.0002 | 0.0728 | 0.0009 | 0.0568 |
| Trial pit 3-leach-River-11 | 0.0024 | 0.1545 | 0.0074 | 0.0097 | 0.0424 | 0.0001 | 0.0003 | 0.0794 | 0.0005 | 0.0493 |
| Trial pit 3-leach-River-12 | 0.0012 | 0.1292 | 0.0079 | 0.0158 | 0.0333 | 0.0003 | 0.0005 | 0.0678 | 0.0003 | 0.0247 |
| **Minimum** | 0.0009 | 0.0738 | 0.0068 | 0.0078 | 0.0243 | 0.0001 | 0.0003 | 0.0646 | 0.0002 | 0.0247 |
| **Maximum** | 0.0029 | 0.1545 | 0.0108 | 0.0158 | 0.0536 | 0.0003 | 0.0006 | 0.0916 | 0.0009 | 0.0568 |
| **Range** | 0.0020 | 0.0808 | 0.0039 | 0.0080 | 0.0294 | 0.0002 | 0.0003 | 0.0270 | 0.0007 | 0.0321 |
| **Median** | 0.0019 | 0.1273 | 0.0078 | 0.0097 | 0.0403 | 0.0001 | 0.0003 | 0.0716 | 0.0003 | 0.0455 |
| **Artificial seawater leachant** | | | | | | | | | | |
| Trial pit 3-leach-Sea-1 | 0.0023 | 4.9258 | 0.0515 | 0.0856 | 0.1319 | 0.0008 | 0.0177 | 0.3672 | 0.0068 | 0.3138 |
| Trial pit 3-leach-Sea-2 | 0.0010 | 4.4401 | 0.0308 | 0.0745 | 0.1326 | 0.0007 | 0.0061 | 0.3943 | 0.0022 | 0.2911 |
| Trial pit 3-leach-Sea-3 | 0.0010 | 4.7481 | 0.0271 | 0.0532 | 0.1023 | 0.0008 | 0.0075 | 0.4272 | 0.0063 | 0.3432 |
| Trial pit 3-leach-Sea-4 | 0.0008 | 5.7756 | 0.0209 | 0.0543 | 0.1261 | 0.0008 | 0.0066 | 0.4060 | 0.0025 | 0.3896 |
| Trial pit 3-leach-Sea-5 | 0.0026 | 5.6428 | 0.0248 | 0.0457 | 0.1609 | 0.0008 | 0.0103 | 0.4767 | 0.0080 | 0.4330 |
| Trial pit 3-leach-Sea-6 | 0.0013 | 4.9591 | 0.0210 | 0.0357 | 0.1033 | 0.0007 | 0.0077 | 0.4158 | 0.0031 | 0.3215 |
| Trial pit 3-leach-Sea-7 | 0.0005 | 0.9077 | 0.0062 | 0.0116 | 0.0257 | 0.0003 | 0.0018 | 0.0752 | 0.0013 | 0.0720 |
| Trial pit 3-leach-Sea-8 | 0.0026 | 8.7527 | 0.0426 | 0.0790 | 0.2872 | 0.0012 | 0.0195 | 0.8060 | 0.0081 | 0.5992 |
| Trial pit 3-leach-Sea-9 | 0.0020 | 5.3184 | 0.0184 | 0.0387 | 0.1558 | 0.0007 | 0.0077 | 0.3908 | 0.0073 | 0.3243 |
| Trial pit 3-leach-Sea-10 | 0.0012 | 5.0449 | 0.0242 | 0.0399 | 0.1058 | 0.0006 | 0.0072 | 0.3736 | 0.0025 | 0.3097 |
| Trial pit 3-leach-Sea-11 | 0.0020 | 5.3577 | 0.0187 | 0.0400 | 0.1193 | 0.0006 | 0.0058 | 0.3477 | 0.0041 | 0.2978 |
| Trial pit 3-leach-Sea-12 | 0.0005 | 0.0284 | 0.0011 | 0.0063 | 0.0075 | 0.0002 | 0.0006 | 0.0104 | 0.0003 | 0.0218 |
| **Minimum** | 0.0005 | 0.0284 | 0.0011 | 0.0063 | 0.0075 | 0.0002 | 0.0006 | 0.0104 | 0.0003 | 0.0218 |
| **Maximum** | 0.0026 | 8.7527 | 0.0515 | 0.0856 | 0.2872 | 0.0012 | 0.0196 | 0.8060 | 0.0081 | 0.5992 |
| **Range** | 0.0022 | 8.7243 | 0.0504 | 0.0793 | 0.2797 | 0.0010 | 0.0189 | 0.7956 | 0.0078 | 0.5774 |
| **Median** | 0.0013 | 5.0020 | 0.0226 | 0.0428 | 0.1227 | 0.0007 | 0.0074 | 0.3926 | 0.0036 | 0.3177 |

**Table A.37: Mass of metal released (µg) per kilogram of matrix material - Leigh Marshes**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 5-leach-River-1 | 51.00 | 1.32 | 4.26 | 1.70 | 20.47 | 34.11 | 7.52 | 63.40 | 1.42 | 487.23 |
| Trial pit 5-leach-River-2 | 39.11 | 2.53 | 4.91 | 3.11 | 22.81 | 60.85 | 6.00 | 87.53 | 3.70 | 977.56 |
| Trial pit 5-leach-River-3 | 112.55 | 2.34 | 4.46 | 4.65 | 20.02 | 52.76 | 2.72 | 64.91 | 2.70 | 1098.49 |
| Trial pit 5-leach-River-4 | 71.33 | 3.55 | 5.25 | 7.31 | 27.01 | 41.76 | 0.48 | 69.91 | 4.11 | 1239.64 |
| Trial pit 5-leach-River-5 | 88.65 | 2.51 | 4.52 | 4.87 | 36.10 | 34.78 | 0.21 | 71.65 | 4.06 | 1391.27 |
| Trial pit 5-leach-River-6 | 59.19 | 1.95 | 4.29 | 2.66 | 17.46 | 46.42 | 0.00 | 42.22 | 6.05 | 565.41 |
| Trial pit 5-leach-River-7 | 131.62 | 2.26 | 4.56 | 3.54 | 16.48 | 54.30 | 1.04 | 59.58 | 3.22 | 1025.73 |
| Trial pit 5-leach-River-8 | 97.99 | 2.54 | 4.43 | 3.83 | 24.37 | 49.29 | 0.65 | 60.25 | 5.95 | 1077.56 |
| Trial pit 5-leach-River-9 | 187.49 | 2.49 | 4.69 | 5.47 | 22.11 | 51.34 | 1.82 | 70.12 | 9.19 | 1268.97 |
| Trial pit 5-leach-River-10 | 28.42 | 1.65 | 1.89 | 2.18 | 8.85 | 21.93 | 1.66 | 25.19 | 1.82 | 446.54 |
| Trial pit 5-leach-River-11 | 330.99 | 2.93 | 4.69 | 4.83 | 40.18 | 53.22 | 2.16 | 71.00 | 39.22 | 1529.18 |
| Trial pit 5-leach-River-12 | 90.25 | 2.47 | 4.99 | 3.68 | 26.86 | 51.55 | 0.67 | 76.07 | 4.39 | 1461.91 |
| Trial pit 5-leach-Sea-1 | 288.12 | 77.24 | 15.83 | 15.67 | 233.39 | 224.00 | 117.72 | 350.19 | 1036.28 | 6269.77 |
| Trial pit 5-leach-Sea-2 | 149.00 | 70.90 | 10.95 | 17.50 | 121.67 | 230.18 | 58.16 | 301.91 | 241.23 | 5406.27 |
| Trial pit 5-leach-Sea-3 | 102.23 | 51.49 | 7.29 | 15.13 | 60.47 | 154.35 | 42.97 | 207.27 | 177.20 | 4438.12 |
| Trial pit 5-leach-Sea-4 | 238.33 | 73.52 | 10.54 | 15.67 | 137.60 | 211.86 | 54.33 | 299.86 | 274.10 | 5932.54 |
| Trial pit 5-leach-Sea-5 | 117.48 | 72.15 | 9.98 | 20.60 | 132.78 | 208.68 | 59.91 | 314.78 | 87.05 | 5447.00 |
| Trial pit 5-leach-Sea-6 | 320.39 | 70.96 | 11.29 | 14.90 | 149.27 | 209.12 | 59.41 | 315.18 | 473.87 | 5838.39 |
| Trial pit 5-leach-Sea-7 | 315.02 | 70.83 | 10.84 | 19.55 | 132.47 | 194.77 | 62.57 | 300.03 | 225.32 | 5384.65 |
| Trial pit 5-leach-Sea-8 | 168.31 | 72.33 | 12.29 | 18.63 | 119.02 | 204.09 | 72.47 | 308.88 | 433.89 | 6249.67 |
| Trial pit 5-leach-Sea-9 | 217.26 | 69.12 | 9.61 | 18.51 | 99.25 | 197.38 | 56.95 | 289.31 | 184.16 | 5301.75 |
| Trial pit 5-leach-Sea-10 | 304.92 | 72.88 | 10.55 | 14.16 | 121.71 | 197.59 | 45.74 | 275.86 | 236.42 | 5374.47 |
| Trial pit 5-leach-Sea-11 | 150.99 | 62.24 | 9.94 | 15.63 | 114.60 | 185.20 | 47.06 | 266.48 | 125.87 | 4179.62 |
| Trial pit 5-leach-Sea-12 | 303.86 | 74.62 | 10.85 | 19.49 | 127.21 | 192.85 | 51.71 | 312.16 | 251.80 | 5032.89 |

**Table A.38: Mass of metal released (µg) per kilogram of matrix material - Hadleigh Marsh**

| Leachate name | Al | Cd | Co | Cr | Cu | Fe | Mn | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Trial pit 3-leach-River-1 | 163.82 | 2.23 | 1.67 | 5.69 | 40.68 | 44.22 | 2.07 | 39.38 | 2.08 | 631.30 |
| Trial pit 3-leach-River-2 | 158.06 | 1.77 | 1.47 | 7.06 | 44.32 | 48.77 | 1.52 | 40.27 | 1.79 | 598.23 |
| Trial pit 3-leach-River-3 | 132.41 | 1.57 | 1.46 | 5.45 | 49.64 | 63.13 | 1.63 | 39.49 | 1.32 | 670.95 |
| Trial pit 3-leach-River-4 | 323.26 | 2.26 | 1.83 | 5.82 | 45.19 | 58.38 | 1.28 | 42.79 | 4.37 | 762.05 |
| Trial pit 3-leach-River-5 | 104.51 | 2.60 | 1.90 | 6.83 | 42.28 | 51.76 | 1.38 | 39.53 | 1.48 | 631.71 |
| Trial pit 3-leach-River-6 | 279.99 | 2.63 | 1.52 | 6.28 | 55.15 | 54.43 | 2.01 | 51.20 | 3.43 | 806.26 |
| Trial pit 3-leach-River-7 | 121.46 | 1.42 | 1.46 | 7.12 | 47.06 | 64.19 | 1.97 | 39.40 | 1.56 | 432.79 |
| Trial pit 3-leach-River-8 | 317.55 | 1.77 | 1.34 | 7.44 | 45.30 | 64.57 | 1.92 | 34.61 | 1.37 | 707.63 |
| Trial pit 3-leach-River-9 | 283.91 | 2.55 | 1.38 | 6.12 | 52.31 | 47.39 | 1.49 | 41.91 | 4.02 | 826.96 |
| Trial pit 3-leach-River-10 | 288.96 | 1.79 | 1.16 | 5.33 | 49.41 | 49.10 | 1.05 | 36.07 | 3.41 | 628.98 |
| Trial pit 3-leach-River-11 | 297.40 | 2.65 | 1.35 | 6.65 | 44.53 | 56.72 | 1.37 | 47.88 | 3.14 | 740.96 |
| Trial pit 3-leach-River-12 | 145.86 | 2.25 | 1.67 | 12.30 | 47.70 | 114.56 | 2.97 | 42.01 | 2.52 | 447.61 |
| Trial pit 3-leach-Sea-1 | 258.01 | 92.73 | 9.88 | 54.22 | 248.13 | 293.92 | 91.68 | 215.05 | 50.28 | 5316.63 |
| Trial pit 3-leach-Sea-2 | 119.48 | 77.67 | 5.17 | 43.58 | 160.50 | 269.87 | 29.66 | 194.72 | 11.76 | 3673.76 |
| Trial pit 3-leach-Sea-3 | 115.05 | 86.41 | 4.83 | 32.68 | 128.93 | 286.83 | 36.59 | 224.38 | 47.00 | 5155.40 |
| Trial pit 3-leach-Sea-4 | 88.26 | 94.43 | 3.28 | 30.21 | 118.86 | 264.92 | 28.95 | 187.50 | 12.92 | 4905.77 |
| Trial pit 3-leach-Sea-5 | 304.26 | 102.91 | 4.25 | 28.39 | 193.16 | 276.39 | 48.83 | 239.26 | 49.21 | 6769.32 |
| Trial pit 3-leach-Sea-6 | 153.38 | 97.34 | 3.90 | 22.66 | 134.61 | 258.83 | 39.31 | 226.67 | 20.18 | 4987.78 |
| Trial pit 3-leach-Sea-7 | 62.88 | 17.37 | 1.17 | 7.89 | 31.41 | 112.25 | 8.77 | 41.63 | 8.24 | 1239.09 |
| Trial pit 3-leach-Sea-8 | 288.10 | 166.47 | 6.85 | 41.27 | 261.10 | 411.53 | 84.38 | 378.96 | 43.64 | 8061.28 |
| Trial pit 3-leach-Sea-9 | 235.87 | 92.18 | 2.83 | 20.73 | 156.10 | 227.31 | 32.13 | 189.25 | 30.19 | 3590.18 |
| Trial pit 3-leach-Sea-10 | 145.91 | 93.78 | 4.15 | 23.21 | 150.85 | 228.51 | 32.18 | 197.43 | 12.75 | 4457.80 |
| Trial pit 3-leach-Sea-11 | 240.14 | 81.14 | 3.06 | 22.32 | 115.76 | 221.50 | 25.38 | 167.29 | 19.96 | 3793.82 |
| Trial pit 3-leach-Sea-12 | 54.47 | 0.47 | 0.19 | 3.92 | 10.95 | 86.21 | 2.89 | 5.43 | 1.57 | 349.49 |